1988

Hydrogeology of the Hillsboro landfill, Hillsboro, North Dakota

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HYDROGEOLOGY OF THE HILLSBORO LANDFILL, HILLSBORO, NORTH DAKOTA

by
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Bachelor of Science,
University of Wisconsin-Platteville, 1986

A Thesis

Submitted to the Graduate Faculty

of the

University of North Dakota

in partial fulfillment of the requirements

for the degree of

Master of Science

Grand Forks, North Dakota

December
1988

This thesis submitted by Jeffrey D. Maletzke in partial fulfillment of the requirements for the Degree of Master of Science from the University of North Dakota has been read by the Faculty Advisory Committee under whom the work has been done, and is hereby approved.

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This thesis meets the standards for appearance and conforms to the style and format requirements of the Graduate School of the University of North Dakota, and is hereby approved.

Dean of the Graduate School

Permission

Title: HYDROGEOLOGY OF THE HILLSBORO LANDFILL, HILLSBORO, NORTH DAKOTA

Department: GEOLOGY

Degree: MASTER OF SCIENCE

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ACKNOWLEDGMENTS

I would like to thank my advisory committee chairman, Dr. John Reid, for his advice, criticism, and guidance throughout this study. Dr. Reid exhibited genuine concern for my progress and the successful completion of my thesis work. I would also like to thank committee member, Edward Murphy, of the North Dakota Geological Society for introducing me to the project and for his invaluable assistance in every aspect of this project. In addition, sincere thanks is extended to committee members, Dr. Alan Cvancara and Dr. Gerald Groenewold for their constructive criticism and professional advice.

The North Dakota Geological Survey furnished the drill rig and materials for piezometer installation. They also graciously allowed me the use of their facilities and equipment. A special thanks to David Lechner for his assistance during piezometer installation and earth resistivity surveying. I am indebted to Ken Dorsher for his drafting expertise, the results of which appear in this report.

Appreciation is extended to the City of Hillsboro for permitting unrestricted access to the landfill site, for excavating two trenches to enable observation of fractures, and for digging me out of a snow bank one cold February day.

The staff of the North Dakota State Department of

Health is gratefully acknowledged. Special thanks are conveyed to Steve Tillotson.

I wish to thank my parents and family. They often asked how my "paper" was going and I was secure in the knowledge that I always had their love and support. Special gratitude is expressed to my wife, Penny, who put her own career goals on hold so that I could further mine.

This report is dedicated to my brother Steven. A Master's thesis was not part of the good Lord's plan for Steve. I hope that other aspects of his life may be as fulfilling for him as completion of my degree was for me.

ABSTRACT

Landfills are the primary means for land disposal of solid wastes, and as such, they represent potential sources of groundwater contamination. This potential for contamination is exemplified by the Hillsboro landfill, which was emplaced above the Hillsboro aquifer, within permeable surface materials, and under shallow water-table conditions. Burial of refuse in trenches 15-feet (4.57 m) deep ensure that at least portions of the refuse are below the water-table, which varied from 5.4 to 13.2 feet (1.7 to 4.0 m) below the surface. Concern over these factors led to the present study.

Subsurface conditions were investigated by electrical, earth-resistivity surveying. Very little contrast was evident in the observed field data and high resistivity values were rare. Quantification of resistivity results and subsequent delineation of a contaminant plume based upon those results proved difficult for these reasons.

Water samples, and consequentially chemical concentration levels, were obtained from piezometers screened in silt and sand at various intervals within the zone of saturation. Although contaminant levels appear to be low, degradation of groundwater beneath the landfill is evident. Most notably, the concentrations of the trace metals arsenic, cadmium, selenium, lead, copper, chromium, iron, and manganese ranged from 3.5 to 20 times more than back-

ground levels.

The configurations of contaminant plumes for most chemical parameters are similar and believed to be the result of longitudinal and transverse dispersion within the saturated zone. These plumes indicate that the buried refuse within landfill trenches is the source of contamination. However, the plume shapes for lead and arsenic probably reflect isolated disposal or surface spills outside of the covered landfill trenches.

X-ray diffraction analyses revealed that, of the clay minerals present, smectite is dominant. Distribution of smectite in the Hillsboro sediments may be effective in attenuating trace metals and may contribute to apparent low levels of released contaminants.

Low precipitation and high evapotranspiration associated with drought conditions experienced during the study period suggest that leachate production was minimal. Leachate generation from percolation probably occurs only during years of above normal precipitation and, even then, only during periods of intense rainfall.

Based upon calculated average linear velocity of groundwater beneath the landfill, groundwater travel time from the northern portion of the buried refuse to the southern limits of the landfill site is nearly 28 years. Given this travel time, the 12-year residence time of the refuse may be inadequate to establish the magnitude of leachate generation.

INTRODUCTION

General Statement

Landfills have been the primary means for the land disposal of solid wastes for many years. In the past, little attention was paid to operation procedures, design criteria, and location of sites within suitable geologic and hydrogeologic settings. In many cases this has resulted in groundwater contamination problems. More recently, an increased awareness by the public and by state and federal regulatory agencies of the threat to groundwater from landfills, has resulted in an effort to upgrade, close, or relocate existing unacceptable sites. The Hillsboro landfill is representative of a site which came under scrutiny because of unacceptable geologic and hydrogeologic conditions. Concern over these factors provided the impetus for this study.

Definition

A logical starting point in any discussion of landfills is a clear and accurate definition of the term "landfill." Taken in the context of this study, "landfill" specifically refers to a sanitary landfill. The North Dakota State Administrative Code, Article 33-20-01-04, defines a sanitary landfill as follows:

> A sanitary landfill is a disposal operation employing an engineered method of disposing of putresicible solid wastes in thin layers, compacting the solid wastes to the smallest practical volume, and applying and compacting cover material at the end of each operating day.

Inherent in such a definition is the consideration of soil characteristics and hydrogeology in landfill site selection and design. It has been estimated that in the past, less than 10 percent of the nation's landfills have been operated within this definition of a sanitary landfill (Water Well Journal, 1974, p. 41). In contrast to sanitary landfills, "landfill" may refer to "any land area dedicated or abandoned to the deposit of urban solid waste regardless of how it is operated or whether or not a subsurface excavation is actually involved" (Water Well Journal, 1974, p. 41).

Significance of Landfills

The significance of landfills lies in their use as the primary means for land disposal of solid wastes, and, as a consequence, in their potential as sources of groundwater contamination. The threat of groundwater contamination has increased concern about the manner in which solid wastes

are collected and disposed. Thus, quantification and characterization of this waste is relevant to a discussion of landfills.

The Environmental Protection Agency (EPA) identified 16,416 landfills in the United States in 1986. Of these landfills, 9,284 (57%) were municipal landfills -- landfills which receive primarily household refuse and non-hazardous commercial waste. West Virginia, Pennsylvania, and Texas reported the largest number of landfills per state (Environmental Protection Agency, 1986a).

Perhaps the best source of information on municipal solid waste is "Characterization of Municipal Solid Waste in the United States, 1960 to 2000" (Environmental Protection Agency, 1986b). This study reports that about 76 million tons (17273 kg) of municipal solid waste were discarded in 1960. The volume of waste rose to approximately 133 million tons (30227 kg) in 1984, and is projected to reach nearly 159 million tons (36136 kg) by the year 2000.

The 133 million tons (30227 kg) of municipal solid waste generated and discarded in 1984 is equivalent to 3.0 pounds (1.4 kg) per person per day (Environmental Protecttion Agency, 1986b). The characterization study (Environmental Protection Agency, 1986b) estimated that of these 133 million tons (30227 kg), 126.5 million tons (28750 kg) were disposed of in landfills. In addition, the study determined that as a national average, over 50 percent of

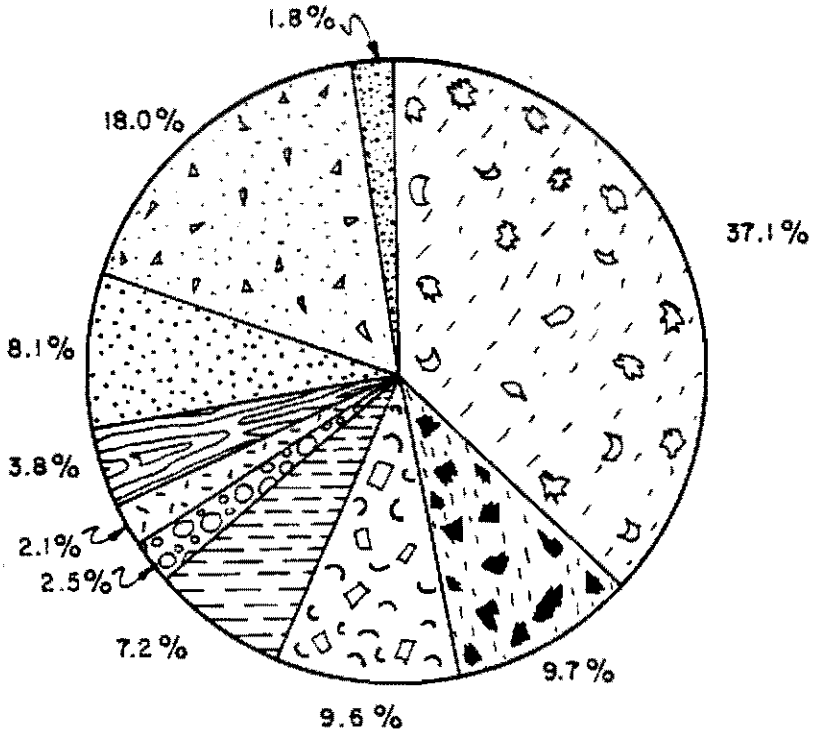
municipal solid waste is composed of paper and yard wastes; almost 40 percent is metals, food wastes, and plastics; and 10 percent is wood, rubber and leather, textiles, and miscellaneous inorganics (Figure 1).

Trends in Landfill Use

Land disposal of solid wastes has been practiced for many years. In the past, uncontrolled dumping and the open-burning dump were common. Over the past 30 years, however, the open-burning dump has gradually been replaced by the sanitary landfill (Cartwright, Griffin, & Gilkeson, 1976). A more recent trend has been a reduction in the number of operating landfills throughout the United States. This is reflective of tougher state and federal environmental regulations, increased public awareness of groundwater problems, and the limited capacity problems of older landfills. The result is increased volumes of refuse at fewer sites, notably near the urban centers (Cartwright, Griffin, & Gilkeson, 1976).

In light of public opposition, finding suitable land on which to place a landfill, tougher permit requirements, and rising costs, the siting of new landfills poses significant difficulties for many states (Environmental Protection Agency, 1986a). In addition, developing alternatives to landfills poses difficulties because of the time, cost, and technology involved. Both incineration and

Figure 1. Materials discarded into sanitary municipal landfills in 1984, by percent of total (after Environmental Protection Agency, 1986b, p. 1-9).



PAPER

GLASS

METALS

PLASTICS

RUBBER & LEATHER

TEXTILES

WOOD

FOOD WASTES

YARD WASTES

MISC.
INORGANIC WASTES

resource recovery may represent viable alternatives and may eventually reduce the volume of solid wastes requiring disposal. However, since even the most complete and effective systems of resource recovery leave up to 80 percent of the original solid waste as residue (Garland & Mosher, 1975), these alternatives do not represent a solution for the immediate capacity problems. Land disposal will likely continue to be the primary disposal method.

Landfills in North Dakota

In North Dakota, landfills are the primary method of disposal of nonhazardous solid wastes. Currently the state has 97 municipal sanitary landfill facilities (North Dakota State Department of Health, 1988). North Dakota reflects the national trend in that it, too, is experiencing a trend toward regional landfilling. It is estimated that there will be only 30 to 40 landfills in the state within the next 5 to 10 years (Tillotson, 1988).

In North Dakota, municipal solid wastes are generated at the rate of approximately four pounds (1.8 kg) per person per day. Thus, North Dakota's 684,920 people produce and dispose of nearly 375,000 tons (85.2 kg) of refuse per year (Tillotson, 1988).

Because there are no hazardous waste disposal facilities in North Dakota, it is the nonhazardous waste

disposal facilities that pose the greatest threat to groundwater in North Dakota. It should be noted however, that household amounts of hazardous waste are disposed of in the nonhazardous waste disposal facilities. Municipal sanitary landfills comprise the largest group of nonhazardous solid waste disposal facilities (North Dakota State Department of Health, 1988). Of the 97 existing municipal sanitary landfills in North Dakota, 41 have been identified since 1980 to have some risk of groundwater contamination. In addition, 18 landfills have been closed, 15 are scheduled to be closed, and 8 will be upgraded (Tillotson, 1988).

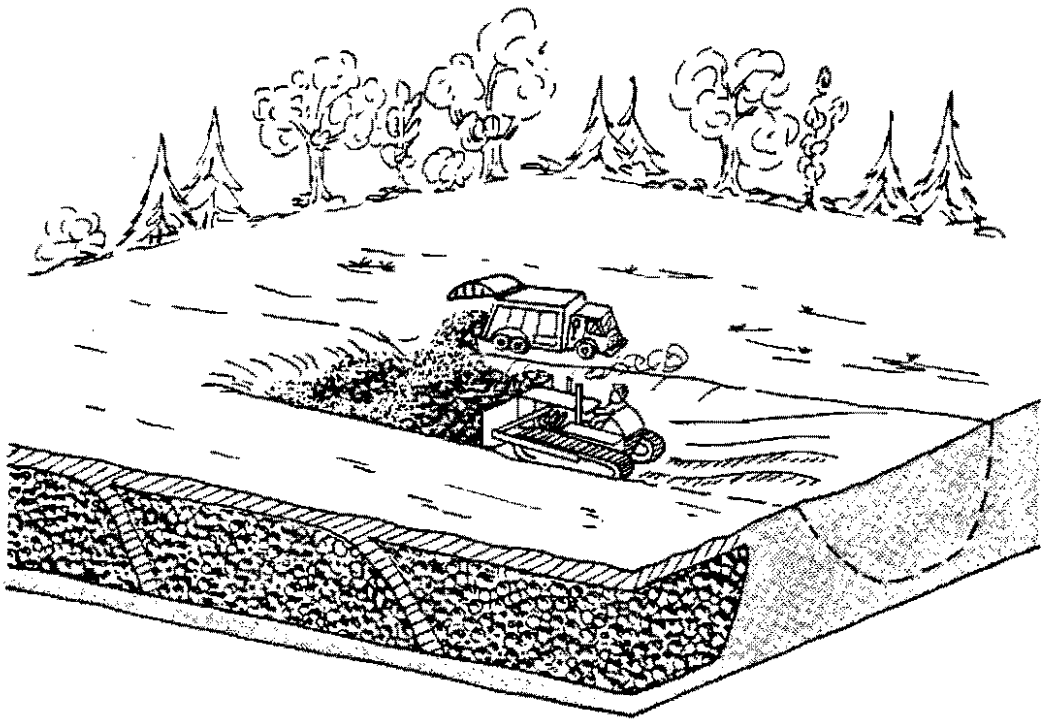
The most common technique utilized in sanitary landfill operation in North Dakota is the trench/fill method. This method involves excavation of a narrow trench, emplacement of refuse, and burial by soil from the excavation (Figure 2).

North Dakota Landfill Regulations

The North Dakota State Administrative Code, Article 33-20, contains the regulations governing landfill operation in this State. These rules embrace the definition of a sanitary landfill and outline procedures for the excavation, compaction and covering of waste, types of waste disposal permitted, and permit applications and renewals.

It is the responsibility of the North Dakota State

Figure 2. The trench/fill method of landfill operation.



REFUSE FILL

Department of Health, Division of Waste Management and Special Studies (NDSDH), to ensure compliance with these regulations. Landfill facilities are aerially inspected at least four times a year by the State Department of Health. The major areas of landfill operational noncompliance include uncontrolled access, open burning, lack of proper compaction and cover, unconfined disposal, and lack of proper construction and operation (North Dakota State Department of Health, 1988).

With an increased awareness for the potential threat to groundwater and surface water from landfills, the NDSDH has concentrated on siting new landfills in areas that are geologically and hydrologically acceptable for waste disposal. This has been especially true since 1980. During this time there has been increased emphasis placed on upgrading, closing, or relocating existing unacceptable sites (North Dakota State Department of Health, 1988).

Project Inception

As early as 1977, Alan Kehew, during a land disposal site inventory for the NDSDH and the North Dakota Geological Survey (NDGS), concluded that the Hillsboro landfill was unacceptable for present use. Kehew (1977) further recommended that instrumentation be installed and a monitoring study initiated.

Attention was refocused on the Hillsboro landfill by

both the NDGS and the NDSDH in 1982. This attention was a direct result of the review of the City of Hillboro's permit renewal application and subsequent concern over the location of the landfill above a major aquifer (Figure 3). In addition to its location, the Hillsboro Landfill was deemed unacceptable because of the high permeability of surface materials and the high water table there.

During May 11 through May 21, 1987, Ecology and Environment, Inc., under the direction of the Region VIII EPA, installed and sampled four monitoring wells, collected soil from the landfill, and collected water and sediment samples from the Goose River (Ecology and Environment, Inc., 1987a). The specific objectives of the sampling were to characterize the waste at the Hillsboro landfill and to determine if a release of contaminants had occurred. Samples were analyzed for metals, volatile organics, base/neutral/acid extractables, pesticides, and polychlorinated biphenyls (PCB). The results obtained for both organic and inorganic contaminants indicated concentrations below detection limits or at or below background values in each case. It was concluded that no contaminant release had occurred at the Hillsboro landfill (Ecology and Environment, Inc., 1987b).

Because of the geologic and hydrogeologic conditions previously cited, the NDSDH felt that the findings of Ecology and Environment, Inc. were inconclusive and that additional investigation was warranted. Thus, the present

study of the Hillsboro landfill was initiated upon the suggestion of the NDSDH, as part of a cooperative project with the NDGS.

Purpose of the Study

The purpose of this study is to determine whether or not the buried refuse in the Hillsboro landfill has undergone leaching, and, if so, to determine the extent and character of the leachate. In addition, this study seeks to investigate geologic, hydrologic, and chemical controls influencing the movement or attenuation of the leachate, and to define any possible threat to the Hillsboro Aquifer.

Location and History of the Hillsboro Landfill

The Hillsboro landfill is located in central Traill County, approximately 3 miles (1.86 km) northwest of the City of Hillsboro (Figures 3 and 4). The landfill site is in the NE1/4 of the SW1/4 of Section 24, T146N, R51W. The landfill has served farm residences and the municipality of Hillsboro. The current population of Hillsboro is 1600 (City of Hillsboro Auditor's Office, 1988).

Operation of the Hillsboro landfill involved the trench/fill method with landfilling proceeding from west to east. The trenches, or landfill cells, are in the northern half of the 11 acre (4.5 hectare) site. NDSDH data

Figure 3. Location of the study site and its relation to the Hillsboro aquifer (after Bluemle, 1967).

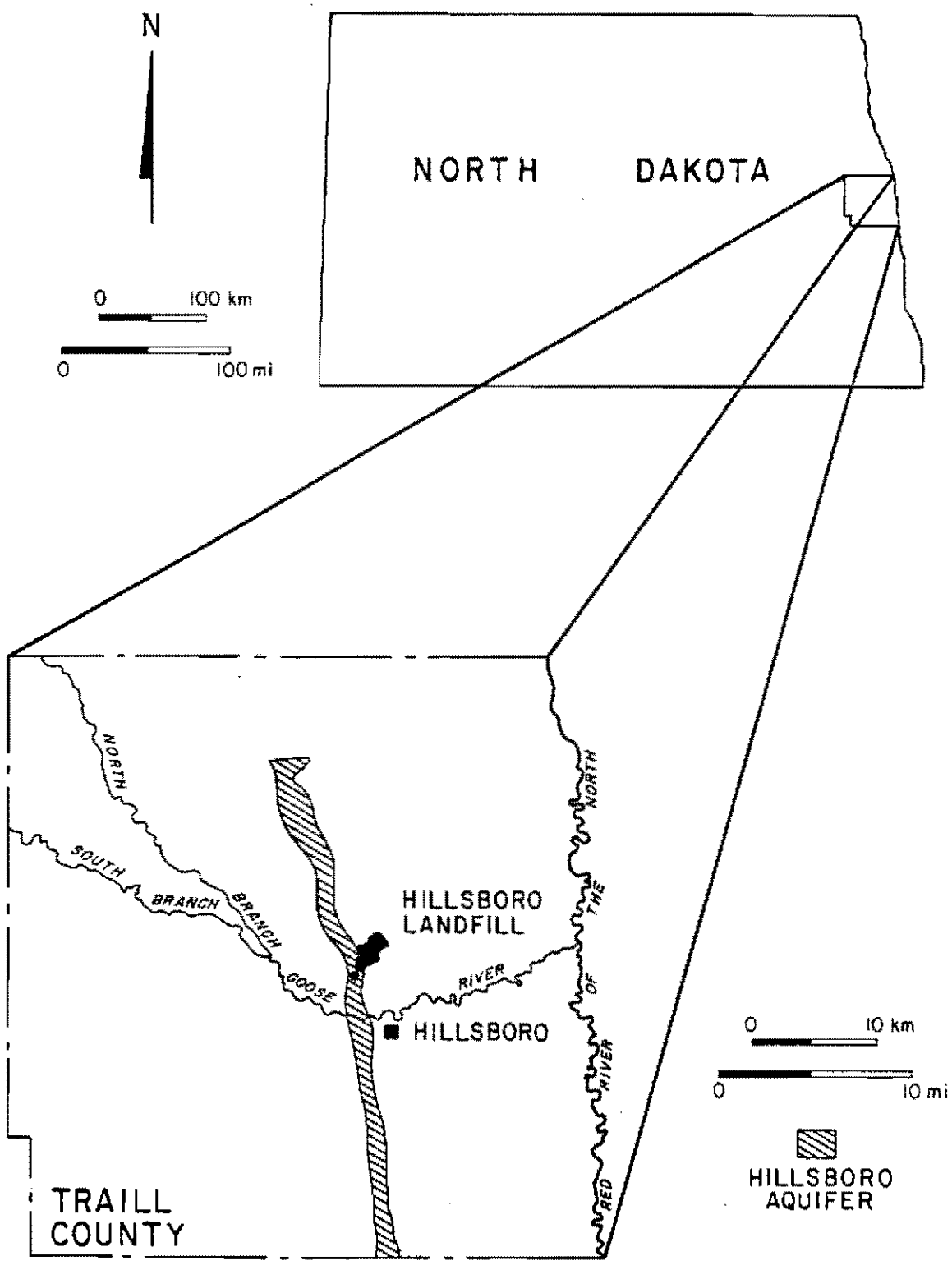
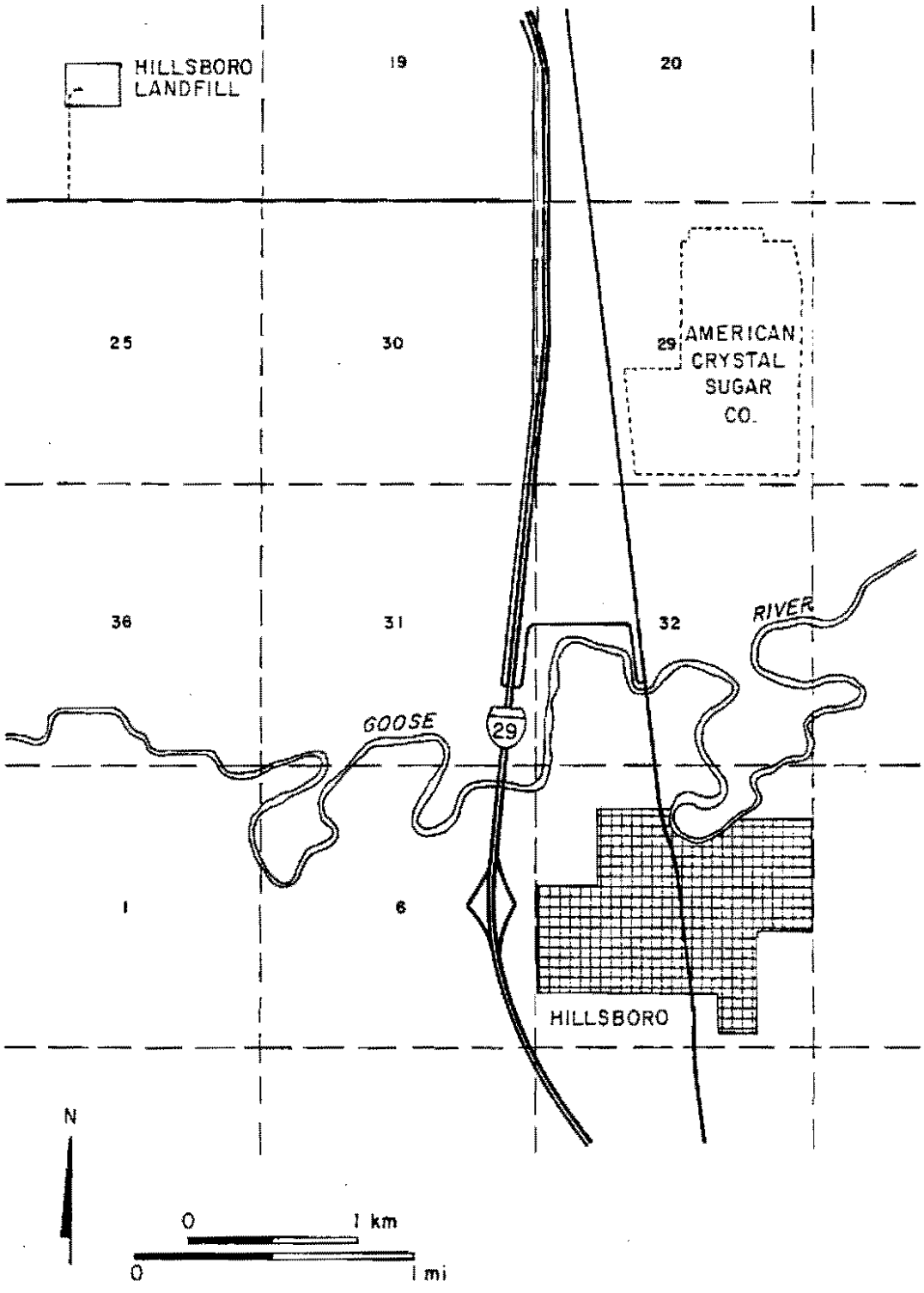

N
NORTH DAKOTA
0 100 km
0 100 mi
NORTH BRANCH GOOSE RIVER
SOUTH BRANCH GOOSE RIVER
NORTH FORK RED RIVER
HILLSBORO LANDFILL
HILLSBORO
TRAILL COUNTY
0 10 km
0 10 mi
HILLSBORO AQUIFER

Figure 4. Location of the Hillsboro landfill in the NE1/4 of the SW1/4 of Section 24, T.146N., R.51W.


HILLSBORO
LANDFILL
19
20
25
30
29
AMERICAN
CRYSTAL
SUGAR
CO.
36
31
32
RIVER
GOOSE
29
1
6
HILLSBORO
N
0
1 km
0
1 mi

indicate that a series of three trenches, approximately 40 feet (12.2 m) wide by 300 feet (91.4 m) long, and 15 feet (4.6 m) deep, were successively excavated and filled. Those records also indicate that the landfill cells were unlined and numerous inspection reports reveal that the site was often operated as an open dump. The reports repeatedly cite the lack of daily or routine cover, and when applied, the lack of sufficient cover (6 inch (15 cm) minimum thickness required). Numerous photos taken in conjunction with the NDSDH aerial inspections often show uncovered garbage in direct contact with standing water in the trenches.

The Hillsboro landfill is owned and operated by the City of Hillsboro. Landfilling operations began in 1976 (North Dakota State Department of Health Records). The landfill was designated and designed for municipal refuse. However, the NDSDH has documented the acceptance of lead acetate from the American Crystal Sugar Company and such farming waste as herbicide containers.

The easternmost trench on the landfill site was filled in September of 1987 and the site is now closed as a municipal landfill. As part of the closure plan, the existing trenches were to be capped with two feet of cover, to include eight inches (20 cm) of clay. Since August of 1987, Hillsboro has been transporting and disposing of its garbage in the Grand Forks landfill, 45 miles (72.4 km) to the north.

The Hillsboro site is currently classified as an inert waste disposal site. As such it can accept construction materials and brush and tree trimmings. A large burn pile consisting of such material is located in the southeast portion of the site. As recently as the fall and winter of 1987-88, scrap iron was stockpiled in the northeast portion of the site. This scrap pile has subsequently been removed. Presently a large pile of old tires occupies the northwest portion of the landfill, and a number of partially buried herbicide containers can be observed in the extreme northwest corner of the site.

Climate

The climate of the study area is continental, subhumid (Ruffner, 1985). The average annual precipitation recorded by the U.S. Weather Bureau at Hillsboro is 19.82 inches (495 mm). Typically, the highest monthly precipitation occurs during June, 3.73 inches (93 mm) on average (National Oceanic and Atmospheric Administration, 1988). The average annual snowfall is between 30 and 35 inches (750 and 875 mm) (Ruffner, 1985). The mean annual temperature is 41.1 degrees Fahrenheit (5.1°C) (National Oceanic and Atmospheric Administration, 1988).

Regional Geology

Traill County is located on the extreme eastern edge of the Williston Basin and all but the southwest corner of the county lies within the glacial Lake Agassiz plain. With the exception of the southwest corner of the county, which is underlain mainly by glacial till, the sediment underlying the surface to a depth of 50 feet (15.2 m) or more is generally silty lacustrine clay. These strata overlie Cretaceous sands and shales. In parts of eastern Traill County however, the sedimentary rocks are absent and the glacial drift directly overlies Precambrian igneous and metamorphic rocks of the Canadian Shield (Bluemle, 1967).

Within the broad, flat plain formerly occupied by glacial Lake Agassiz, the principal landforms include deltas, underflow fans, beaches, and compaction ridges. The deltas, composed of sandy silt, rise as much as 50 feet (15.2 m) above the lake plain in western Traill County. Beaches, several feet (0.91 m) high and composed of sand and gravel, cross the deltas and the lake plain in a general north-south direction (Jensen and Klausing, 1971). As mapped by Clayton (1980), compaction ridges are the result of differential compaction of coarser fluvial sediments within finer lacustrine sediments. The sand and gravel of the fluvial sediments compacts less than the surrounding silty clays of the lacustrine sediments, thereby forming a low ridge.

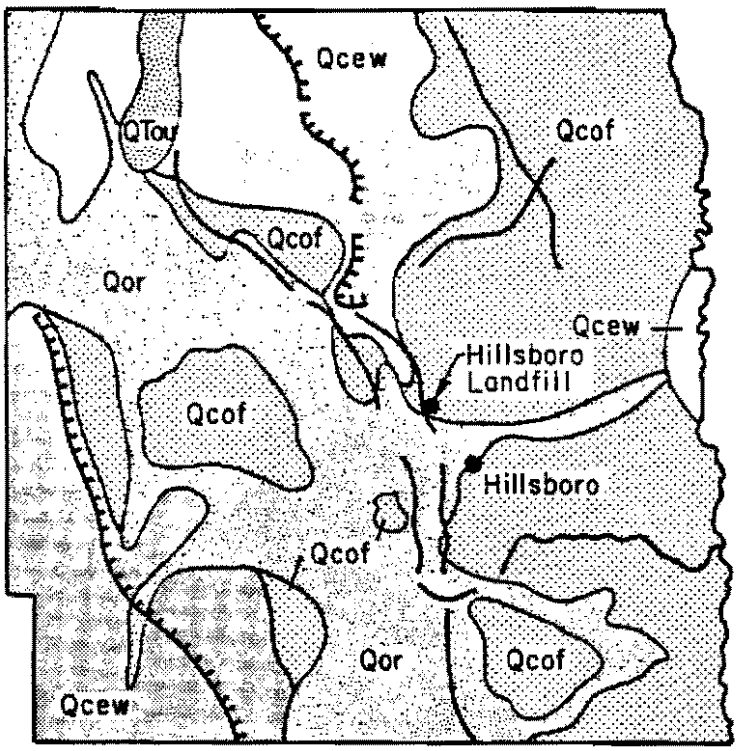
The map, "Maximum Extent and Major Features of Lake Agassiz," published recently (Teller and Clayton, 1983), coupled with the history of Lake Agassiz reported in other papers in that volume, offer detailed insight into the geology of the surface of Traill County and the Lake Agassiz basin. In addition, the complexity of surficial sediment within the Lake Agassiz basin is shown on the Geologic Map of North Dakota (Clayton, 1980). The location and extent of various glacial deposits in Traill County are shown in Figure 5.

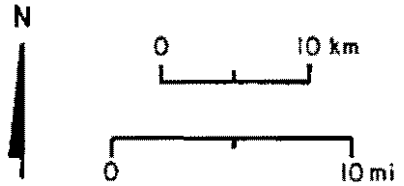
Regional Hydrogeology

Simpson (1929) first discussed the groundwater in Traill County. A more detailed study, focusing on the groundwater resources in the vicinity of Hillsboro, North Dakota, was undertaken by Jensen and Bradley (1963). More recently (Jensen and Klausing, 1971) described the location and extent of groundwater sources in all of Traill County, as well as the chemical quality of the water available from each source.

The major aquifers in Traill County are associated with either the bedrock formations or the glacial drift and include the Dakota Aquifer, the Hillsboro Aquifer, the Galesburg Aquifer, the Elk Valley Aquifer, and the Belmont Aquifer. In addition, where the Lake Agassiz beach deposits exist below the water table they form aquifers

Figure 5. Simplified geologic map of Traill County, North Dakota (after Clayton, 1980).





Qor — RIVER SEDIMENT (HOLOCENE)

QTou — WINDBLOWN SAND

Qcof — PROGLACIAL-LAKE SEDIMENT

Qcew — WAVE-ERODED GLACIAL SEDIMENT

COMPACTION RIDGE

CREST OF BEACH RIDGE

that yield small supplies of water. Locally, small sand and gravel lenses and layers in the till may also serve as sources of water for small public supplies (Jensen and Klausing, 1971).

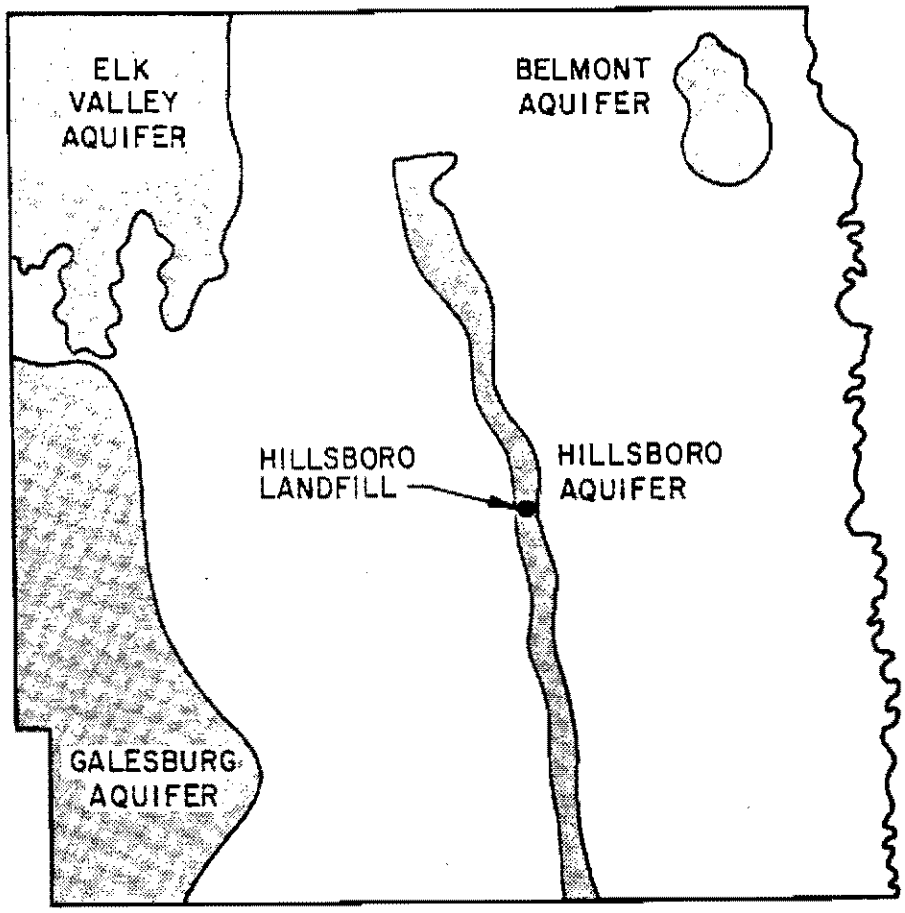
The Dakota Aquifer, a bedrock aquifer, is highly mineralized and chiefly used for rural domestic and livestock purposes. The deposits of sand and gravel associated with the glacial drift contain the most useful groundwater supplies. About 70 percent of the wells in Traill County obtain water from the Hillsboro, Galesburg, Elk Valley, and Belmont Aquifers (Jensen and Klausing, 1971) (Figure 6). Holocene alluvium is present along stream channels, but the deposits are thin and small, and are not a source of groundwater for wells.

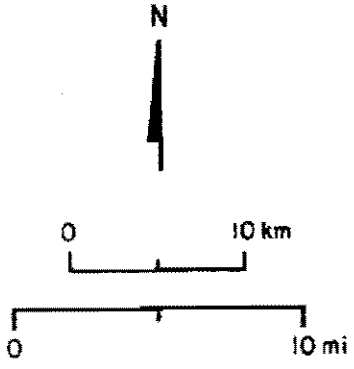
Groundwater Recharge

Groundwater recharge is defined as the entry of water into the saturated zone and the associated flow away from the water table within the saturated zone. In contrast, infiltration refers to the entry of water into the soil and the associated flow away from the ground surface within the unsaturated zone (Freeze and Cherry, 1979, p.211).

Groundwater recharge is influenced by many variables which contribute to the complexity of the process. As part of the water cycle, water derived from precipitation can infiltrate the subsurface and begin the process of

Figure 6. Major glacial drift aquifers in Traill County, North Dakota (after Bluemle, 1967).


ELK
VALLEY
AQUIFER
BELMONT
AQUIFER
HILLSBORO
LANDFILL
HILLSBORO
AQUIFER
GALESBURG
AQUIFER


N
0
10 km
0
10 mi

groundwater recharge. Generally, only a small portion of total annual precipitation is available as recharge. The remaining water is lost through runoff, evaporation, or transpiration. The critical factors controlling the rate and amount of infiltration and subsequent recharge include the permeability and antecedent moisture content of the sediment and the amount of water available (Rehm and others, 1982).

Groundwater recharge can occur throughout the year. The majority of groundwater recharge occurs during the months of March and April; between April and October evaporation and transpiration losses are high and recharge may be negligible (Rehm and others, 1982). This is consistent with water level fluctuations reported by Jensen and Klausing (1971) for the major aquifers in Traill County associated with the glacial drift; increased water levels were noted between the end of March and the end of April, whereas yearly lows were observed in January through March.

During the winter months infiltration may be restricted or prevented due to the formation of frost and the fact that precipitation is locked up in the form of snow and ice. The accumulated snow is often not available for groundwater recharge until the end of March.

Hydraulic Conductivity

Hydraulic conductivity is a quantitative measurement

of the ability of a material to transmit fluid. Within the saturated zone, the permeability of the sediment and the density and viscosity of the fluid flowing through it directly influence this measurement.

Values of hydraulic conductivity vary over a wide range, reflecting an equally wide range of geological materials. However, an order-of-magnitude knowledge of hydraulic conductivity can be very useful. Typically, hydraulic conductivity is high for sand and gravel and low for clay and most rock.

The hydraulic conductivity of the fine to coarse sand comprising the Hillsboro Aquifer is on the order of 3.28 X 10-5 ft/s (10-5 m/s) (Jensen and Klausing, 1971) and the sand and gravel of the beach deposits is on the order of 3.28 X 10-3 to 3.28 X 10-4 ft/s (10-3 to 10-4 m/s) (Ecology and Environment, Inc., 1987a). The silty clay of the glacio-lacustrine sediments of the Lake Agassiz plain are likely on the order of 3.28 X 10-5 to 3.28 X 10-8 ft/s (10-5 to 10-8 m/s) (Freeze and Cherry, 1979, p.29).

Water Quality

All natural water contains dissolved solids. As water infiltrates the subsurface its chemical character is altered upon the dissolution or partial dissolution of minerals. The type and amount of dissolved solids in groundwater is primarily dependent upon the solubility and

type of minerals in contact with groundwater, the length of time the water is in contact with them, the organic acid and CO_2 content of the soil, the rate of groundwater flow, and the initial chemical composition of the water.

The quality of water and the suitability of water for various uses is reflected in the kind and quantity of dissolved solids. Commonly cited chemical indicators of groundwater contamination, applicable to most hydrogeological settings, include total dissolved solids, ammonium, chloride, bicarbonate, iron, potassium, sulfate, and nitrate (Saar and Braids, 1983; Andersen and Dornbush, 1967; Jensen and Klausing, 1971).

The predominant water types for the glacial drift aquifers in Traill County are calcium bicarbonate, sodium bicarbonate, calcium sulfate, and sodium chloride. Dissolved solids average approximately 1400 ppm and sulfate content averages nearly 385 ppm (Jensen and Klausing, 1971). A more detailed discussion of the chemical analysis of groundwater in Traill County can be found in Jensen and Klausing (1971).

PREVIOUS WORK

An extensive library search and a nationwide computer database search through GEOREF, Water Resources Abstracts, and National Technical Information Service (NTIS) produced nearly 500 references dealing with various aspects of groundwater contamination from municipal landfills. This section serves as a literature review. Some of the sources presented will be referenced in greater detail in subsequent parts of this study. In addition, an attempt is made to focus on how changing attitudes and subsequent laws and regulatory action have influenced the direction and scope of groundwater contamination investigations with regard to landfills. The sampling activities at the Hillsboro landfill, conducted by Ecology and Environment, Inc., under the direction of the EPA, are also discussed.

A review of the literature reveals both a regional bias with regard to the principal studies concerned with groundwater pollution from sanitary landfills and a variation in groundwater pollution laws from state to state. Zanoni (1972) provided an excellent summary of important early studies (1950 - 1970), in which he noted that California, South Dakota, and Illinois were leaders in initiating and conducting research in the area of groundwater pollution from the land disposal of solid wastes. Other states at that time which were becoming more active included Maryland, Pennsylvania, and Wisconsin. Zanoni

(1972) also included a survey of state regulations regarding groundwater pollution from landfill operations. His results indicated highly variable codes and guidelines; a few states had no published codes or guidelines, whereas others, like California, required detailed classification of wastes and disposal sites to guard against groundwater pollution. It is safe to conclude that those states, in which most of the research was conducted, were more keenly aware of the potential for groundwater pollution from landfills and accordingly enacted the most stringent regulations for the approval of landfill sites.

With the advent of the Clean Water Act (CWA), passed in 1972, and the Safe Drinking Water Act (SDWA), passed in 1974, a renewed emphasis was placed on groundwater pollution research. Accordingly, the literature during this time (1970s) reflects an emphasis on the evaluation of hydrogeologic and geologic criteria in landfill site selection and design, and on the design and implementation of effective monitoring programs.

Again, the regional bias was evident, as many of the most useful studies were by the Illinois State Geological Survey. Hughs (1972) stressed consideration of hydrologic and geologic factors in landfill site selection and design. Several important and often-cited studies concerned the attenuation properties of clay minerals (Griffin and Shimp, 1975; and Griffin and others, 1976 and 1977). These studies concluded that the attenuation capabilities of

clays are an effective means of removing most toxic constituents found in municipal leachates and that optimal clay percentages used in conjunction with hydraulic conductivity information can provide for properly designed landfill liners. Cartwright, Griffin, and Gilkeson (1976) designed laboratory studies to predict the behavior of landfill leachate in porous material. They maintained that leachate migration rates are predictable if the composition of both the leachate and porous media are known. Classes of solid waste sites in Illinois and their hydrogeologic environments were presented by Clark (1975).

In addition to the states previously mentioned, several other relevant studies have been conducted on groundwater pollution from sanitary landfills. Palmquist and Sendlein (1975a) evaluated the hydrology of a refuse site in alluvium to determine the size and shape of zones of contaminated groundwater. They concluded that flood-plain sites may be desirable as landfill sites because of the predictability of the size and shape of contaminated groundwater zones, the tendency for floodplains to be groundwater discharge sites , and the resulting low concentrations of leachate produced in a high groundwater flow environment. In a separate study, Palmquist and Sendlein (1975b) related the topography and hydrogeology of a landfill site to the shape of the resulting zone of contaminated groundwater. They suggested that the size of the site must be sufficient to contain both the waste and

the resulting zone of contamination.

Kunkle and Shade (1976) discussd some of the problems and objectives of designing and implementing groundwater monitoring programs at sanitary landfills. Underscoring the importance of landfill design, Giddings (1977) reviewed the development of a 120-acre (48.6-hectare) sanitary landfill in Pennsylvania which utilized a plastic membrane liner for leachate collection and a backup leachate collection system consisting of groundwater underdrains.

As was the case with the CWA and the SDWA, enactment of the Resource Conservation and Recovery Act (RCRA) in 1976 provided the impetus for continued research on groundwater pollution associated with landfills. In 1979, the EPA promulgated "Criteria for classification of solid waste disposal facilities and practices." The criteria were intended for use in determining which solid waste disposal practices posed contamination and health risks. Landfill sites that violated these criteria were to be classified as open dumps rather than sanitary landfills (Environmental Protection Agency, 1986c). States were urged to prohibit, close, and upgrade open dumps. In conjunction with the introduction of these criteria, the EPA (1986) conducted a census of state non-hazardous waste programs, including landfills. Data on regulations, enforcement, number of landfills, and design and operating characteristics were obtained. The results were presented as the Subtitle D Study Phase I Report (Environmental Protection Agency,

1986c). The Subtitle D census revealed that over 40 states had some location standards or restrictions applicable to municipal landfills, and required groundwater monitoring. However, very few specific design requirements had been implemented by the states. Comparison with the results of Zanoni (1972) indicate a trend of stricter and more encompassing groundwater protection laws. A recent example is Wisconsin's new code requirement, which provides that the interpretation of the geologic and hydrogeologic data in the reports received for a landfill be performed by a qualified hydrogeologist (Gass, 1988, p. 552).

In recent years, much of the emphasis in groundwater investigations has concentrated on the migration of contaminants and on the hydrogeochemical processes at work in landfills. The goal of many of these studies has been to provide a basis for developing an improved methodology for predicting the impact of landfills on groundwater in a variety of geologic and hydrogeologic settings and to provide adequate descriptions of the processes that control contaminant migration in groundwater. One such detailed study is presented as a group of papers in a special issue of the Journal of Hydrology (v.63, no.1/2, May, 1983), in which a large zone of shallow contaminated groundwater in a sand aquifer of an abandoned landfill was examined. As part of this study Dance and Reardon (1982) quantified the effects of cation exchange in order to predict major cation chemical changes during contaminant migration. They

noted that in flow systems characterized by marked changes in pH, only a pH-dependent sorption model can adequately describe ion-sorption reactions. Also, Nicholson, Cherry, and Reardon (1982) drew upon thermodynamic principles for their interpretations of those chemical processes causing the attenuation of contaminants. Similarly, Baedecker and Back (1979) found that degradation of organic matter, redox reactions, and cation-exchange processes were dominant influences on contaminated groundwater from a landfill. As an extension of that study, Baedecker and Apgar (1984) determined that the chemical reactions and processes in contaminated water do not remain constant with time.

Two other aspects of groundwater contamination associated with landfills have come to the forefront of research in recent years. Unsaturated flow of groundwater has been recognized as an important factor in contaminant transport and studies of the occurrence and behavior of organic compounds in contaminated groundwater have been initiated. Johnson and Cartwright (1980) have shown that similar hydrologic and geochemical processes can occur in sanitary landfills in the unsaturated zone as in sites within the zone of saturation. Rapid movement of contaminants in the unsaturated zone have been attributed to capillary forces (Cartwright, 1984).

Many common organic chemicals have been recognized as hazardous and relatively mobile in permeable groundwater systems. The literature acknowledges the complexities of

organic chemical transport in groundwater and the influence this imposes on the design of detection and monitoring systems (Pettyjohn and Hounslow, 1983). Reinhard and others (1984) characterized the organic constituents in landfill leachate plumes and discussed the geochemical, physical, and biological processes that affect their distribution. Chian (1977) conducted a detailed analysis of organics in landfill leachate to assess the attenuation of organics in groundwater. Barker and others (1986) presented an approach to the study of organic contaminant migration at a sanitary landfill site in which they examined the ability of selected trace organics to interact with inorganic contaminants. It has been noted that with regard to organic contaminants, hydrogeologists are in the "process discovery" stage (Barker and others, 1986).

FIELD AND LABORATORY METHODS

Installation of Monitoring Wells

During May 13 - 16, 1987, Ecology and Environment, Inc. installed four shallow monitoring wells (H12, H13, H14, H15) (Figure 7) to obtain groundwater samples for Hazard Ranking System evaluation. Well drilling and formation sampling were provided by Twin Cities Testing of Bismarck. Each monitoring well was drilled with a CME-75 hollow stem auger. Wells were constructed using 2-inch (5.08 cm) inner diameter schedule 80 PVC casing, and two 5-foot (1.52 m) sections of factory slotted .010-inch (0.025 cm) PVC screen. All wells were completed with security casings and locks (Ecology and Environment, Inc., 1987). Typical well construction for these wells is shown in Figure 8.

For the purpose of this study, additional geologic and hydrogeologic information was sought through test drilling and subsequent installation of additional piezometers. During November 13 through December 1, 1987, eleven piezometers (H1 - H11) (Figure 7) were installed in boreholes drilled with the North Dakota Geological Survey's truck-mounted 8-inch (20 cm) hollow stem auger (Mobil B-50).

The piezometer locations were selected to obtain adequate control, based on an inferred direction of groundwater flow to the south. An attempt was made to place a

Figure 7. Map of the Hillsboro landfill. Water sampling instrumentation sites are shown.

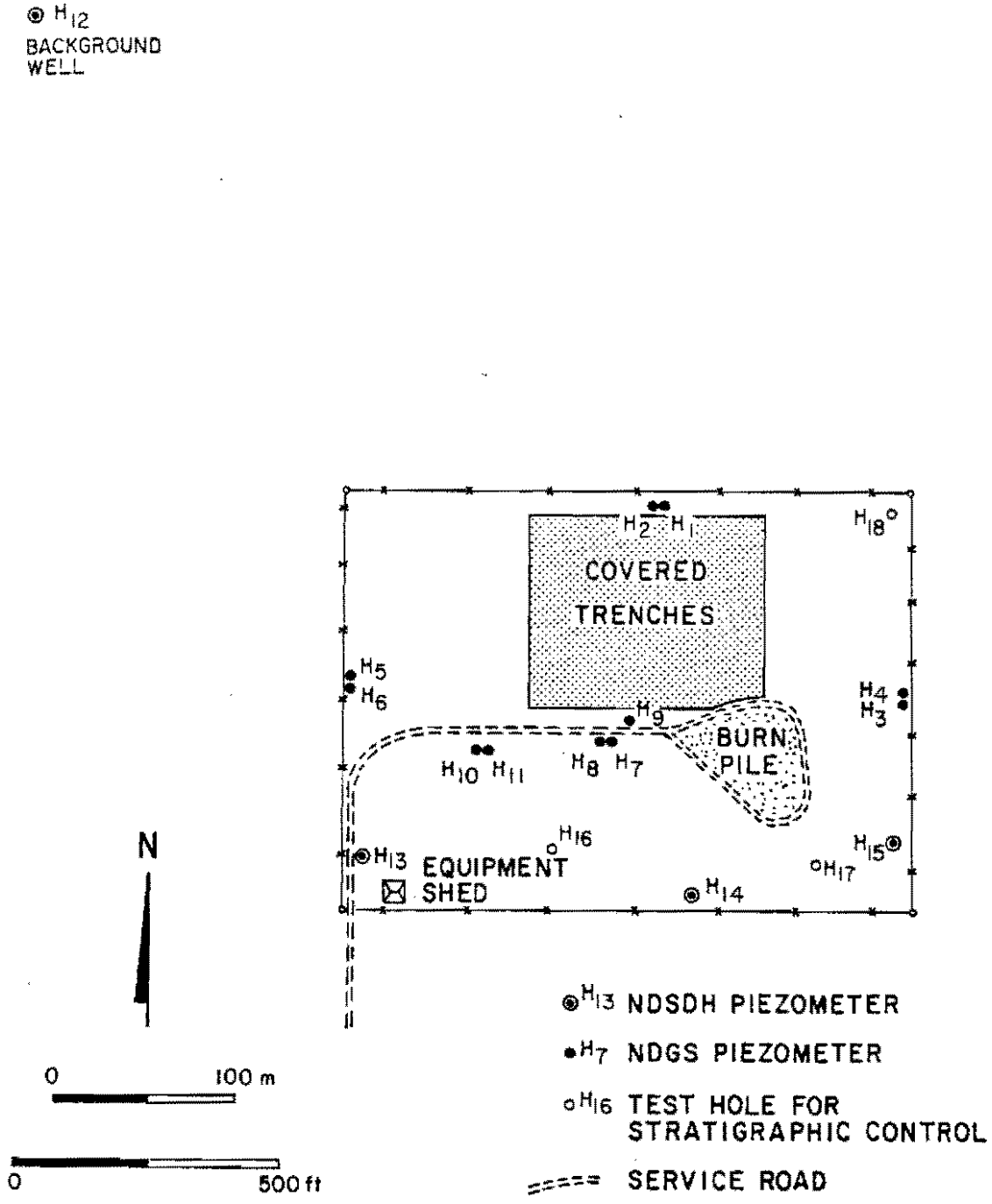
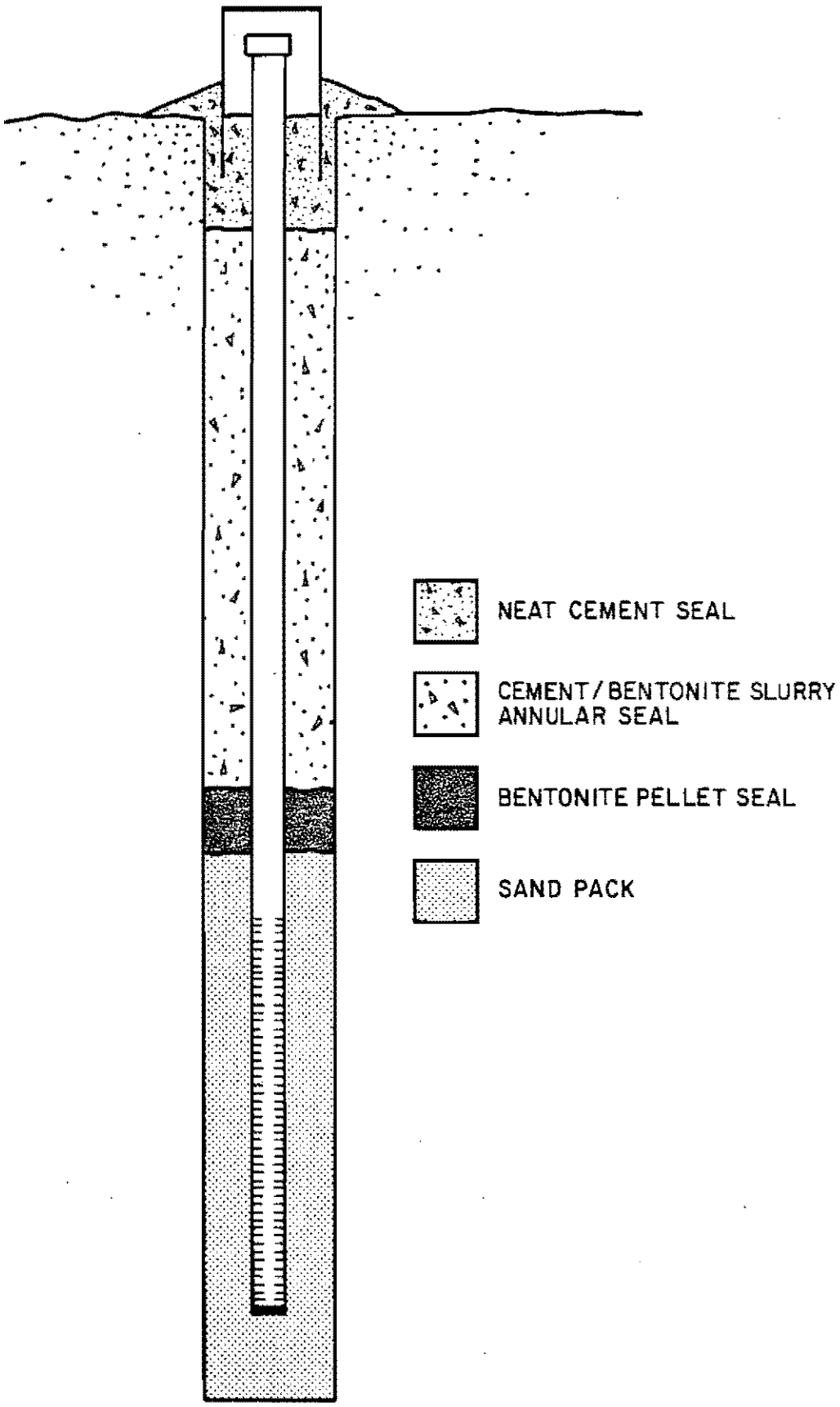

H12
BACKGROUND WELL
H2
H1
H18
COVERED TRENCHES
H5
H6
H4
H3
H9
BURN PILE
H10
H11
H8
H7
H16
H13
EQUIPMENT SHED
H15
H17
H14
N
0
100 m
0
500 ft
H13 NDSDH PIEZOMETER
H7 NDGS PIEZOMETER
H16 TEST HOLE FOR STRATIGRAPHIC CONTROL
SERVICE ROAD

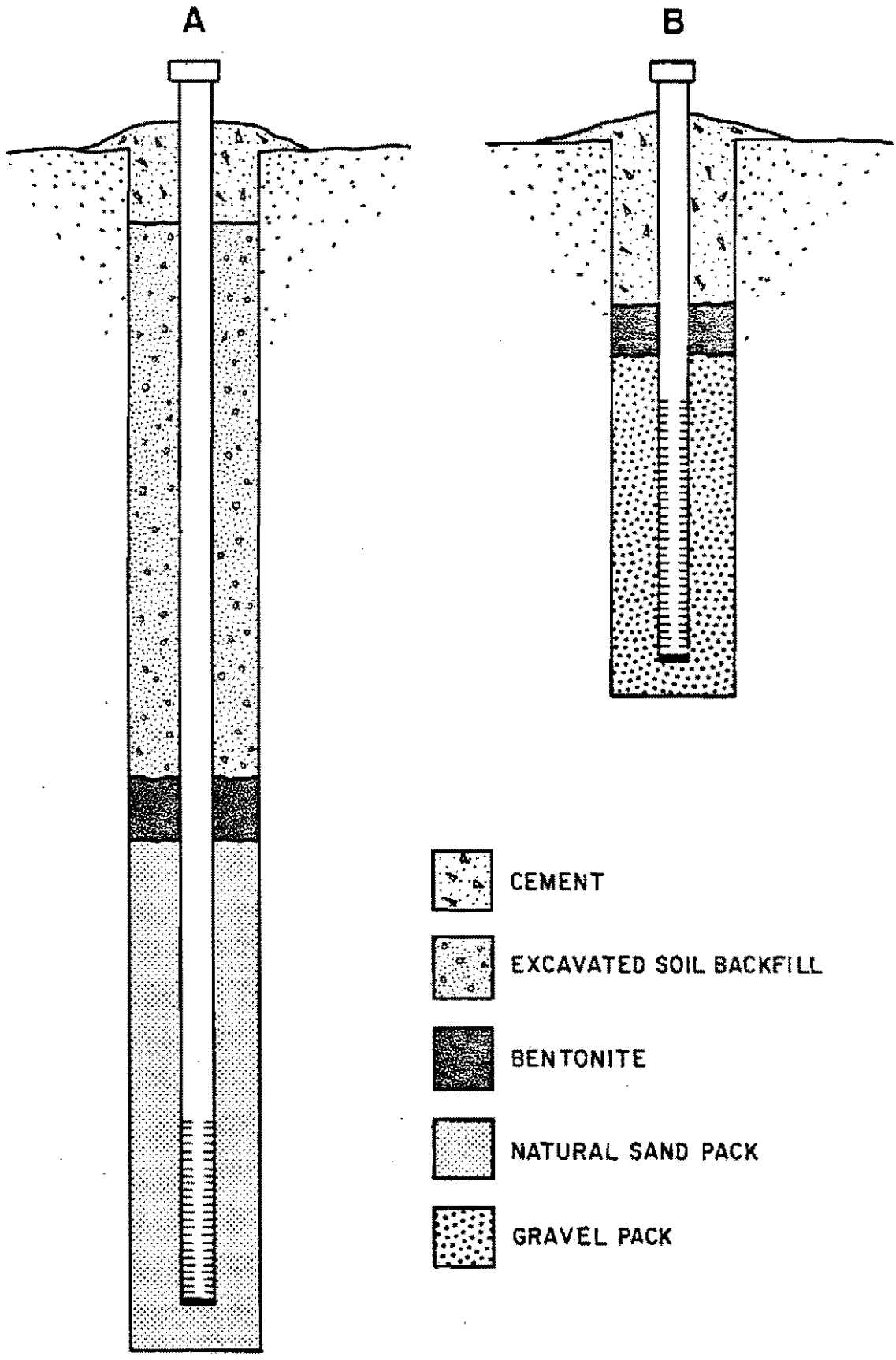
Figure 8. Typical well construction for wells installed by Ecology and Environment, Inc. (after Ecology and Environment, Inc., 1986a).


NEAT CEMENT SEAL
CEMENT/BENTONITE SLURRY ANNULAR SEAL
BENTONITE PELLET SEAL
SAND PACK

number of piezometers as close as possible to the buried refuse, yet avoid drilling directly into the garbage. The piezometers were installed to obtain more precise groundwater data, including water samples for chemical analysis from within the zone of saturation.

The piezometers consist of 2-inch (5.08 cm) diameter schedule 40 PVC casing connected to a 2- or 5-foot (0.61 or 1.52 m) section of preslotted .010-in (0.025 cm) PVC screen. The piezometers were generally nested in pairs to depths of approximately 12 feet (3.66 m) and 32 feet (9.75 m) (Appendix I). One deep piezometer (H9) was screened at a depth of 58 feet (17.7 m). The borehole was drilled to 82 feet (25 m); however, difficulty in retrieving the center bit from the hollow stem auger because it was clogged with sand resulted in collapse of the hole. The shallow piezometers were equipped with a 5-foot (1.52 m) screened section; the deeper piezometers were constructed with 2-foot (0.61 m) screens. Placed in fine to medium sand and silt, the boreholes collapsed after emplacement of these piezometers, thereby forming natural sand packs around the screens. The shallow boreholes generally remained open after the piezometer was in place. Washed pea gravel was placed around the screens. Two different sealing configurations were used. One consisted of a 2-foot (0.61 m) bentonite seal, excavated soil backfill, and a bentonite or cement surface seal (Figure 9A). The other consisted of a 2-foot (0.61 m) layer of bentonite directly

Figure 9. Profiles of the two different sealing configurations used during the installation of; (A) deep piezometers and (B) shallow piezometers.


A
B
CEMENT
EXCAVATED SOIL BACKFILL
BENTONITE
NATURAL SAND PACK
GRAVEL PACK

above a natural sand pack or the washed gravel pack, followed by cement to the surface (Figure 9B). The choice of sealing configuration was not dictated by geologic or hydrogeologic conditions, but rather by economics and the limited amount of cement available for this project.

Collection of Sediment Samples

During the installation of the piezometers, sediment samples were collected for lithologic and stratigraphic information (Appendix II). Representative samples were thereby available for textural analysis. Most of the samples were collected as catch samples (material retrieved from the borehole, representative of an assumed depth interval). The catch samples were described, bagged, and labeled. In addition to the catch samples, 45 feet (13.72 m) of shelby tube sediment samples were taken from two boreholes (H9 and H10). These samples consisted of 2.5-foot (0.762 m) by 3-inch (7.62 cm) sediment cores.

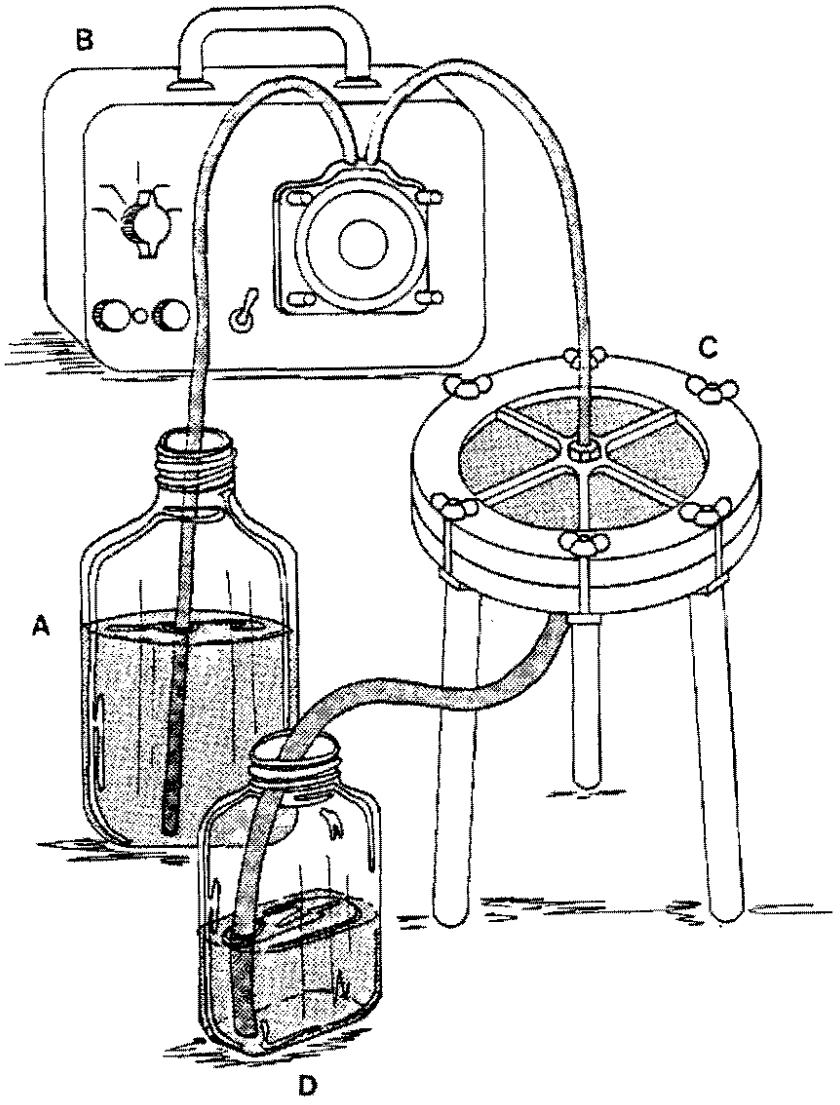
Collection of Water Samples

Water samples were collected in late December, 1987 and in August, 1988. It was originally intended to sample during the spring of 1988; however, because of drought conditions and the absence of a normal recharge event, sampling was delayed. The August sampling did not provide

the recharge conditions sought; however, it was hoped that by preventing further delay that the results of the analyses could be incorporated into this report. However, the results were not received in time for inclusion here.

In an attempt to provide representative samples, the piezometers were bailed dry 24 hours prior to sampling. In the event that the well could not be bailed dry, 3 to 5 well volumes were removed. In addition, at least two well volumes were removed immediately prior to sample collection. A teflon bailer was used to withdraw water from the wells. In order to minimize the oxidized portion of the sample, the first 6 inches (15 cm) of water in the bailer were discarded. Measurement of the temperature, pH, and electrical conductivity of the water samples was performed immediately upon collection. Due to the turbidity of the water samples, they were filtered through prefilters and ultimately through 0.45-micron filters. The filtering was accomplished through the use of a peristaltic pump and filtering apparatus (Figure 10). A one-litre filtered sample was collected for major-ion analysis and a one-half-litre sample was collected for trace metal analysis. Five millilitres of concentrated nitric acid were added to the filtered one-half litre samples to prevent precipitation of trace metals. In order to retard chemical or biological change in the water samples, they were packed in ice-filled coolers during transport to the laboratory. All water samples were analyzed by the North Dakota State Department

Figure 10. Filtering apparatus used in the field. The sample is drawn from the transfer bottle (A), by a peristaltic pump (B) and through the filter (C), to a sample container (D) (after Lindorff and others, 1987, p. 46).


B
C
A
D

of Health and Consolidated Laboratories (NDSDHCL).

For a more detailed description of the equipment used and of the field sampling procedures see Environmental Protection Agency (1980) and Lindorff and others (1987).

Monthly Water Levels

To determine the distribution of hydraulic heads, direction of groundwater flow, and effects of precipitation at the landfill site, monthly water levels were measured from November, 1987 through November, 1988, using a battery-powered water level tape.

Preparation of Base Map

The landfill site was surveyed with plane table and alidade, and a base map was subsequently constructed. Positions and elevations of the monitoring wells were duly recorded. The elevation of the base station was determined from the Hillsboro, North Dakota Quadrangle of the United States Geological Survey, 7.5- minute topographic map series.

Fracture Analysis

In an attempt to evaluate the influence of fractures within the subsurface, two trenches were excavated with a

backhoe. The trenches were dug at right angles to each other in the west-central portion of the landfill, south of the buried refuse. The trenches were approximately 15 feet (4.57 m) long, 15 feet (4.57 m) deep, and 4 feet (1.22 m) wide.

Slug Tests

In situ hydraulic conductivity values were determined by means of single-well response tests. A single-well response test is initiated by causing a change in hydraulic head (water level) in a piezometer. The recovery rate of the water level is then monitored. In this study a solid cylinder, or slug was used to induce a change in hydraulic head. Two slugs designed to raise the water level in a 2-inch (5.08 cm) PVC pipe 3.3 feet (1.0 m) and 1.7 feet (0.5 m) were used. The smaller slug was used when the larger slug could not be lowered into the well due to constrictions or bends in the pipe. After the slug was dropped into the water an electric tape was used to measure the declining water level. The depth of water and the time of measurement were recorded at frequent intervals until the water level had recovered to equilibrium (falling head test). Similarly, as the slug was pulled out of the piezometer, the rate at which the water level rose and ultimately regained equilibrium, was measured (rising head test). Of the 15 piezometers at the Hillsboro site, three

were not tested because there was not enough water to cover the slug.

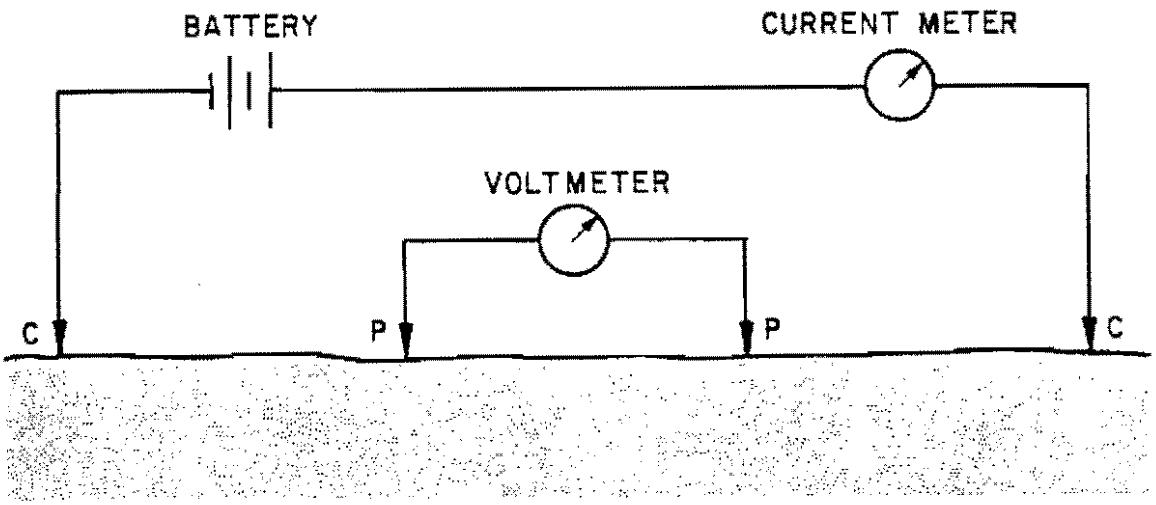
Earth Resistivity Survey

Electrical earth resistivity surveys have been used as a method for defining both variations in stratigraphy (Kehew and Groenewold, 1983; Heigold and others, 1985; Schwartz and McClymont, 1977) and groundwater chemical composition (Stollar and Roux, 1975; Kelly, 1976). A number of studies applied the method to groundwater contamination associated with landfills (Cartwright and McComas, 1968; Klefstad and others, 1975; Cartwright and Sherman, 1972). Electrical resistivity surveying investigates subsurface conditions by passing an electric current into the ground through a pair of current electrodes and measuring resulting voltage difference between a pair of potential electrodes (Figure 11A). A resistivity survey was conducted at the Hillsboro landfill in November, 1987. A Soiltest R-50 Stratameter and R-65 voltmeter were used for this project.

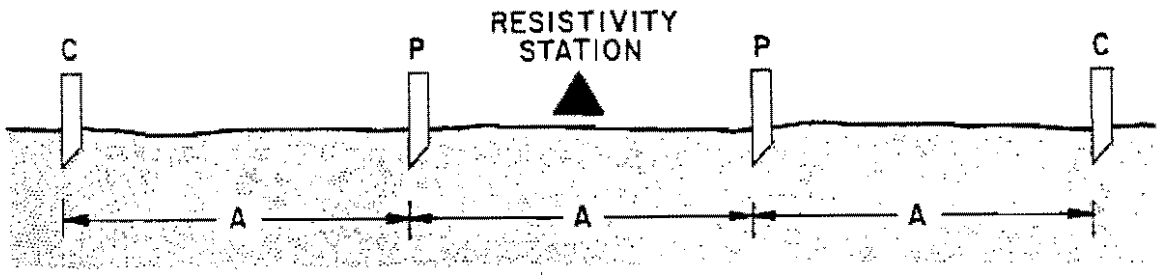
The Wenner electrode configuration, coupled with the Vertical Electrical Sounding (VES) method, were used in the resistivity survey of the landfill (Soiltest, Inc., 1968). The Wenner configuration involves four electrodes equally spaced along a line. The outer electrodes served as current electrodes and the two inner electrodes served as

Figure 11. (A) Configuration of the four electrode array used in the electrical earth resistivity survey. Current was passed through a pair of electrodes (C) and the voltage difference was measured between a pair of potential electrodes (P). (B) The Wenner configuration, as used in this study, involves four electrodes equally spaced along a line.

A


BATTERY
CURRENT METER
VOLTMETER
C
P
P
C

B

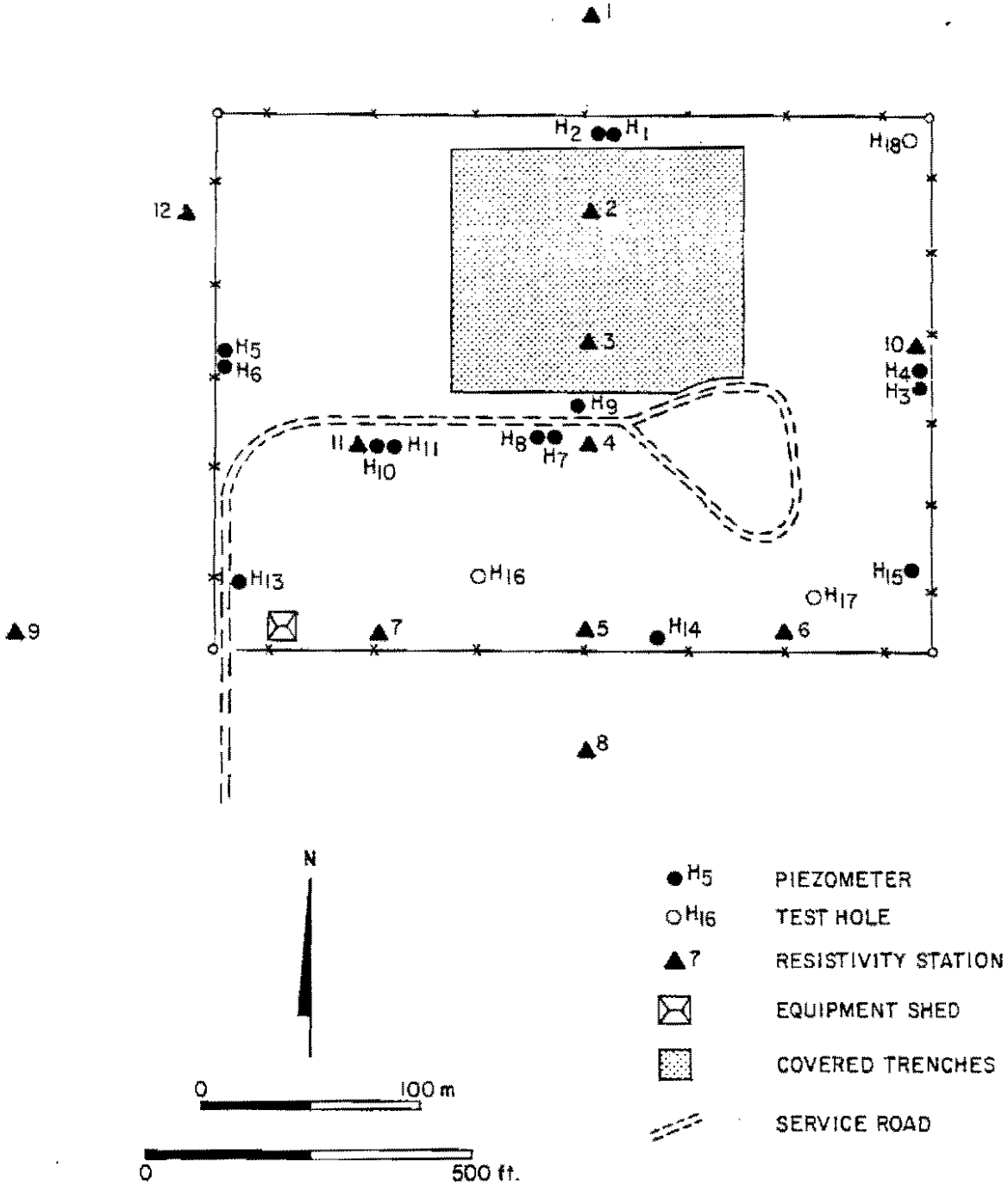

C
P
RESISTIVITY
STATION
P
C
A
A
A

potential receiving electrodes. The distance between adjacent electrodes is designated "A" (Figure 11B). For the VES method, the center of the electrode spread is fixed and the separation of the electrodes is progressively increased. This fixed center is designated as the resistivity station. At each resistivity station (Figure 12), multiple values of apparent resistivity are obtained as the electrode spacing, or A-spacing increases. Electrode spacings of 3, 5, 8, 10, 12, 16, 20, 24, 30, 40, 50, 60, 80, and 100 feet (0.9, 1.5, 2.4, 3.1, 3.7, 4.9, 6.1, 7.3, 9.1, 12.2, 15.2, 18.3, 24.4, and 30.3 m) were used in this study.

Texture Analysis of Sediment Samples

Texture analysis, involving a combination of sieve and hydrometer techniques (Murphy and Kehew, 1984), determined the sand, silt, and clay percentages of sediment samples collected at the Hillsboro landfill. Approximately 45 grams of air-dried sample were weighed and put in a beaker. The sample was soaked overnight in 125 millilitres of four percent Calgon solution (to determine the hydrometer weight of the Calgon, a test cylinder of four percent Calgon solution was prepared (125 ml)). After soaking, the sample was put in a mechanical mixer, stirred with distilled water, and agitated for one or two minutes. It was then decanted into a settling cylinder (1000 ml graduated

Figure 12. Map of the Hillsboro landfill. Both water sampling instrumentation and earth resistivity stations are depicted.


1
H2
H1
H18
12
2
H5
H6
3
10
H4
H3
H9
11
H11
H10
H8
H7
4
H13
H16
H15
H17
9
7
5
H14
6
8
N
0
100 m
0
500 ft.
H5 PIEZOMETER
H16 TEST HOLE
7 RESISTIVITY STATION
EQUIPMENT SHED
COVERED TRENCHES
SERVICE ROAD

cylinder). If any clay balls coated with sand grains were present they were gently broken up with a glass stirring rod and agitated in the mixer with additional distilled water until completely dispersed before being added to the settling cylinder. The cylinder was topped off with distilled water and agitated for about 45 seconds with a perforated rubber stopper attached to an iron rod. Any sand or gravel clinging to the stopper was washed off with distilled water into the soaking beaker and added to the sample during wet sieving. The sample was allowed to settle for approximately two and one-half hours depending on the water (room) temperature (two hours and thirty-three minutes for 22°C). The hydrometer reading was recorded and the test Calgon reading subtracted from it to obtain the clay weight. The sample was then wet-sieved and the sand and gravel was returned to the soaking beaker. The sample was oven-dried at 100°C, and then put on the Ro-Tap mechanical shaker for ten minutes with No. 10 (2 mm), No. 18 (1 mm), and No. 230 (63 micron) sieves. The sand envelopes were weighed during sieving, then filled with the sand and gravel fraction and weighed again, subtracting the envelope weight to obtain the sand and gravel weight. The gravel was subtracted from the original sample weight and the corrected weight was used to calculate the sand, silt, and clay percentages. All weight not accounted for by the gravel, sand, and clay was considered silt.

Hydraulic Conductivity Estimation from Grain-Size Analysis

Hydraulic conductivity values were estimated from grain-size distribution curves using the method of Masch and Denny (1966). Grain-size data, from the results of sieve analysis, were plotted as cumulative percent versus grain-size diameter in ϕ units. The method uses the inclusive standard deviation, σi, where,

$$\sigma i = \frac{d16 - d84}{4} + \frac{d5 - d95}{6.6}$$

The values of d16, d84, d5, and d95 were taken directly from the grain-size gradation curve. As an example, d16 is the grain-size diameter at which 16 percent by weight of the sample is finer and 84 percent is coarser. The median grain size, d50, is taken as the representative diameter. Hydraulic conductivity was subsequently determined from a family of curves (Masch and Denny, 1966, Fig. 8, p. 673), using σi and d50.

Textural determination of hydraulic conductivity provides useful estimates for sediment in the fine sand to gravel range. Accordingly, as determined from the sand, silt, and clay percentages, only samples with a high sand percentage (>50%) were selected for this method. A sequence of Standard U.S. sieves was used, covering the sand-size range of 4ϕ (0.0625 mm) to -1ϕ (2 mm) at 0.50ϕ

intervals.

Hydraulic Conductivity Estimation from Slug Tests

The field recovery data from slug tests were plotted as unrecovered head difference versus time on semi-logarithmic paper, according to the method of Hvorslev (1951). From this plot the basic time lag (To) was measured graphically and the hydraulic conductivity was determined using To and the well dimensions, according to the formula:

$$K = \frac{(R)^2 \ln(L/R)}{2LTo},$$

where K = hydraulic conductivity, L = length of well screen, R = radius of well screen, and To = basic time lag.

Computer Interpretation of Apparent Resistivity Data

Field resistivity data were entered into a computer program developed by Zohdy and Bisdorf (1975) and adapted for PC use (Zohdy and Bisdorf, 1988). The computer program automatically calculates depths and layer resistivities.

In the absence of a PC program for the automatic interpretation of Wenner sounding data, a program designed

to interpret Schlumberger sounding data was modified for this project (Zohdy and Bisdorf, 1988). To interpret Wenner sounding data, the program multiplies each electrode spacing (A) by 3/2. This effectively converts the standardized electrode spacing of the Wenner configuration, AB/3, to that of the Schumberger configuration, AB/2. The program then plots the field curve of apparent resistivity versus electrode spacing, or depth, calculates a best fit curve, and determines depths and layer resistivities. The number of layers interpreted by the program is directly dependent on the number of electrode spacings used in obtaining field resistivity data.

Clay Analysis by X-ray Diffraction

The attenuating ability of clays for the chemical constituents in municipal landfill leachate is dependent on the type of clay (Griffin and others, 1976). Therefore, the types of clay in the sediment samples from the Hillsboro landfill were determined by x-ray diffraction (XRD). As determined from the sand/silt/clay ratios, those samples with the highest clay-size fraction were prepared for XRD analysis. Approximately eight grams of air-dried sample were placed in a 100-millitre beaker. The beaker was filled with distilled water and the sample was stirred, covered, and allowed to disaggregate for 4 to 7 days. After the disaggregation period the sample was agitated for

one minute in order to resuspend the sediment. Stoke's Law was used to compute the time required for the 2-micrometre size fraction to settle to a depth of one centimetre (approximately 45 minutes for 22°C). The <2-micrometre size fraction was then removed by inserting a disposable micropipette into the beaker and drawing off the clay particles with suction. The pipette was allowed to just make contact with the suspension surface, and then lowered to a depth of one centimetre, before suction was applied. The clay suspension, which was drawn off with the pipette, was then put into small glass vials where the clay particles were allowed to settle for at least 72 hours. After settling and thereby concentrating the clay at the bottom of the vial, a micropipette was used to draw off the clay-size fraction. This slurry was mounted on a standard 3.5-inch (8.8 cm) glass optical microscope slide, allowed to dry, and then placed in a desiccator for at least four hours with ethylene glycol in the bottom of the desiccator. This procedure allows the ethylene glycol to replace the water in the smectite, thereby uniformly expanding the [001] lattice plane (Starkey and others, 1984).

Samples H1B, H1D, H5A, H5B, H5C, H5D, H9A, H10C, and H11A (Tables 1 and 2) were analyzed with a Philips (Norelco) model 12045 x-ray diffractometer. The samples were scanned from 3° to 30° two theta at a rate of one degree per minute using a scale factor of 250 and a time constant of one at room humidity. Due to equipment

failure, the remainder of the samples were analyzed on a Philips XRG 3100 x-ray diffractometer under similar conditions. However, the later samples were run at higher killivolt and milliampre settings and were scanned at a rate of one-half degree per minute. Samples H5D, H9A, and H11A were analyzed with both machines in an attempt to access the accuracy of the results from the failed equipment.

Chemical Analysis of Water Samples

The water samples collected at the Hillsboro landfill were analyzed for major ions and trace metals by the North Dakota State Department of Health, Division of Chemistry Laboratory, using the following methodologies:

1. Zinc, copper, barium, iron, manganese, sodium, potassium, calcium, and sulfur were analyzed by emission spectroscopy using a Perkin-Elmer Plasma II inductively coupled plasma emission spectrometer. This system uses two-point background correction and vacuum monochrometers
2. Chromium, arsenic, and selenium were analyzed on a Perkin-Elmer 5100 Atomic Absorption spectrometer using stabilized temperature platform furnace technology and Zieman background correction to control interferences from high chloride content.
3. Lead and cadmium were analyzed on a Perkin-Elmer model

5000/500 atomic absorption spectrophotometer using stabilized temperature platform furnace technology. Conductivity was measured on a Beckman RC19 bridge.

4. Chlorides and nitrates were analyzed on a Lachat flow injection analysis system using colorimetric methods.
5. Alkalinity was determined using a Fisher pH meter.

All analyses were performed using EPA methodology. Spikes (a sample of known composition used to evaluate the accuracy of the analytical techniques used) and duplicates were performed on a minimum of 10% of all samples. Known EPA reference samples were run with all metal analyses (Reetz, 1988).

Computer Evaluation of Leachate Production

In an attempt to better evaluate possible leachate production at the Hillsboro landfill, the Hydrologic Evaluation of Landfill Performance (HELP) computer program was used. The HELP program was developed by the U.S. Army Corps of Engineers Waterways Experiment Station for the EPA. The program is a two-dimensional hydrologic model of water movement across, into, through, and out of landfills. Thus, the effects of precipitation, surface storage, runoff, infiltration, percolation, evaporation, soil moisture storage and lateral drainage are incorporated. Various landfill systems, including combinations of vegetation, cover soils, waste cells, impermeable soil

layers, liners, and covers, may be modeled. The HELP program requires climatologic data, including daily precipitation, mean monthly temperatures, mean monthly insolation, leaf area indices, and winter cover factors. In addition, the various materials contained in the landfill (clay, sand, waste, etc.) and the physical layout of the landfill (size, thickness of layers, slopes, etc.) must be specified. More detailed explanations concerning data requirements, nomenclature, and other fundamental information needed to run the program are presented in the HELP user's guide (Schroeder and others, 1983a).

As with all groundwater modeling programs, there are built-in assumptions and, therefore, limitations. An important limitation of the HELP model is that the actual rainfall intensity, duration, and distribution are not considered. Also, the variables controlling daily evapotranspiration are interpolated from mean monthly values and, as a result, calculated daily values may be quite different from actual daily values. In addition, the program uses several simplifying assumptions and assigns constants and correction factors for several variables. Detailed solution methods for all the modeled hydrologic processes are presented in the HELP documentation (Schroeder and others, 1983b).

The Hillsboro landfill was modeled as a three-layer system consisting of an upper sandy loam layer, a middle layer of waste, and a lower silty loam layer. Surface

cover was entered as good grass and representative values of leaf area index and solar radiation were used. Mean monthly temperatures were based on data from the Hillsboro gauging station from 1950 to 1980. The program was run with several different sets of precipitation data. Initially, daily precipitation values were derived from monthly means during 1950 through 1980. Subsequent applications used monthly precipitation totals during the period of study and, in an attempt to simulate a worst case scenario, the daily precipitation totals at the Hillsboro station in 1953 were used. The annual precipitation at the Hillsboro station in 1953 was nearly 27 inches (67.5 cm), over 6.5 inches (16.3 cm) above normal. It was envisioned that this type of simulation would provide a good contrast when compared to the drought conditions experienced during the period of study. The contrast sought would be representative of a normal recharge event during above average annual precipitation and the negligible recharge experienced during drought conditions.

GEOLOGY OF THE HILLSBORO LANDFILL

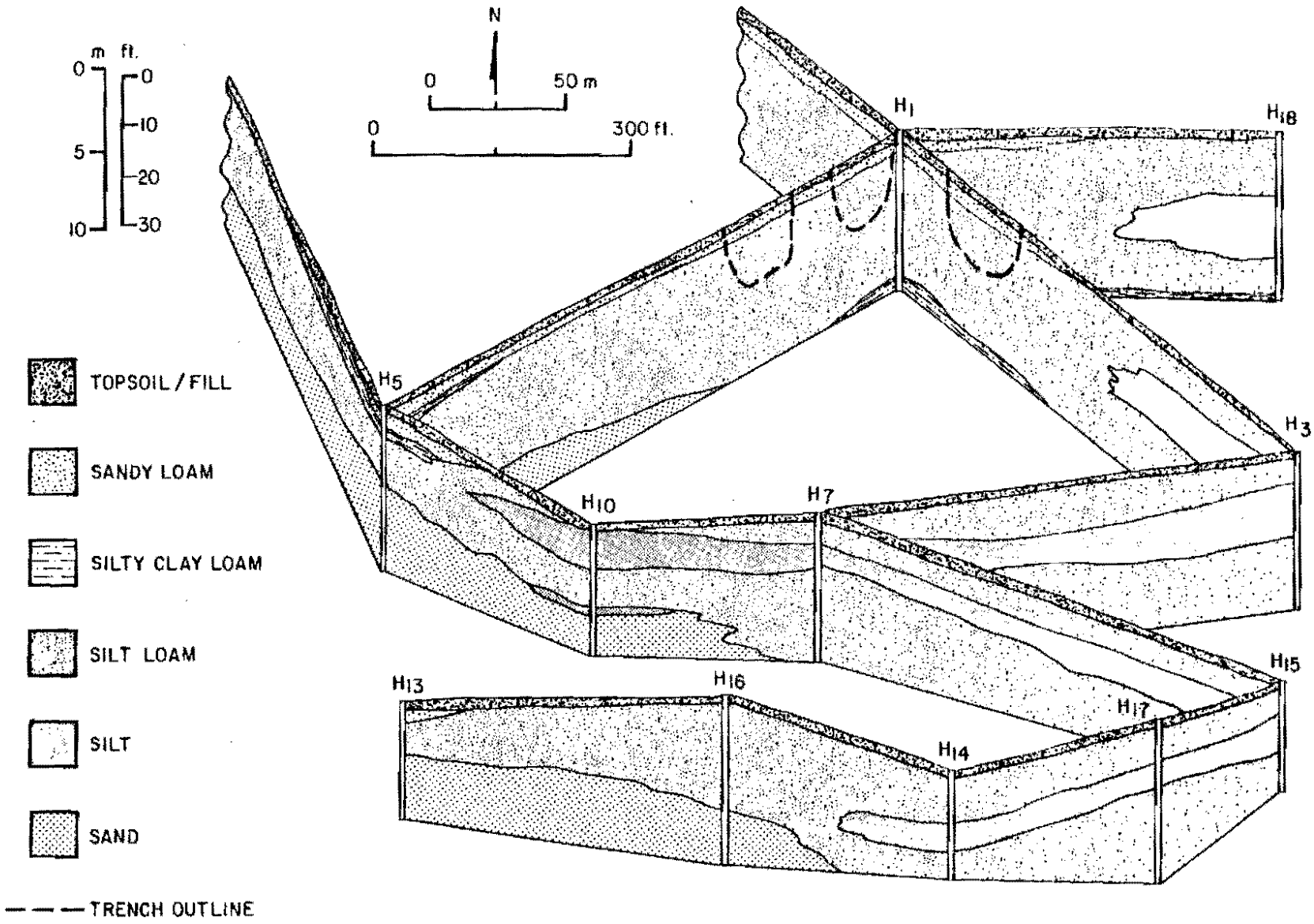
Figure 13 is a geologic fence diagram that illustrates the three-dimensional stratigraphy at the Hillsboro landfill. The individual units are designated according to the U.S. Department of Agriculture textural classification based on sand/silt/clay ratios determined from sediment samples.

The silty clay loam, silt loam, and silt are typical of the sediments common to the Glacial Lake Agassiz plain (Bluemle, 1967). The silt loam is the most pervasive lithology underlying the site. This deposit is uniformly textured, consisting of silt, sand, and small percentages of clay, oxidized to various hues of pale brown (10YR, 6/3) and light yellowish brown (10YR, 6/4). The included sand is generally fine to very fine-grained and well-rounded.

Especially prevalent in the central and eastern portions of the landfill is a layer of silt within the depth interval of approximately 7 to 17 feet (2.13 to 5.18 meters). In places, very fine, close banding of the silty sediments was observed. Within the western extent of both the silt loam and silt deposits, are interbedded, fine-grained sand lenses up to six inches (15 cm) wide and approximately two feet (0.61 m) long.

The distribution of the surficial sandy loam and the lower sand coincide with the eastern margin of a compaction ridge (Figure 5) and the Hillsboro aquifer (Figure 6),

Figure 13. Geologic fence diagram of the Hillsboro landfill. For location of holes, see Figure 12.


m ft.
0 0
10
5 20
10 30
N
0 50 m
0 300 ft.
H1
H18
H5
H3
H10
H7
H15
H13
H16
H17
H14
TOPSOIL / FILL
SANDY LOAM
SILTY CLAY LOAM
SILT LOAM
SILT
SAND
TRENCH OUTLINE

respectively. The sand of the sandy loam is typically fine-grained, well-sorted, and well-rounded, brown (10YR, 5/3) to dark grayish brown (10YR, 4/2). The sand of the Hillsboro aquifer contains very little silt (or clay) and is the coarsest sediment encountered on the site. Deposition of the sediment at the Hillsboro landfill is attributed to accumulation in proglacial lakes and the fluvial influence of drainage into these shallow lakes (Clayton, 1980).

HYDROGEOLOGY OF THE HILLSBORO LANDFILL

The Hillsboro landfill is on the eastern margin of the Hillsboro aquifer (Figure 3). Understandably, concern has arisen over the burial of waste in close proximity to one of Traill County's major aquifers. The aquifer is 1 to 2 miles (1.6 to 3.2 km) wide, and can be traced over about 25 miles (40.2 km) (Jensen and Klausing, 1971). The sand of the aquifer deposit is typically very fine to coarse grained. The hydraulic conductivity of the aquifer as a whole is probably greater than 3.28 X 10-5 ft/s (10-5 m/s) (Jensen and Klausing, 1971). The lower sand encountered in piezometer boreholes H5, H10, and H13 (Figure 13) probably represents the uppermost portion of the aquifer.

During the monitoring period (November, 1987 to August, 1988), the depth to the water table varied from 5.4 to 11.0 feet (1.7 to 3.4 m) below the surface in the northern part of landfill site and from 10.6 to 13.2 feet (3.2 to 4.0 m) below the surface in the southern part of the landfill site. From May through August (1988), the water table beneath the site dropped an average of 4 feet (1.2 m). This rather substantial decline is apparently reflective of the drought conditions that prevailed during the spring and summer of 1988.

Water table maps (Appendix III) indicate that groundwater below the site is flowing to the south-southeast, away from the covered landfill trenches. The gradient of

the water table averages 4.78 X 10-3 ft/ft (1.46 X 10-3 m/m) within the landfill site.

The velocity of groundwater flow is dependent on the hydraulic gradient, porosity, and hydraulic conductivity of the sediments through which it moves. Accordingly, the average linear velocity of the groundwater at the Hillsboro site was calculated using the formula:

$$v = K/n(dh/dl),$$

where v = average linear velocity (m/s), K = hydraulic conductivity of the sediment (m/s), n = sediment porosity, and dh/dl = gradient of water table surface (m/m). The porosity was estimated to be 0.40 for silt and sand (Freeze and Cherry, 1979, p. 37). The average linear velocity calculated from the equation above is 8.59 X 10-7 ft/s (2.62 X 10-7 m/s, or 8.26 m/yr). Hydraulic head values between nested pairs of piezometers (H1 & H2, H3 & H4, H5 & H6, H10 & H11) were used to determine the vertical gradient under the site. The average vertical gradient is 1.36 X 10-1 ft/ft (4.14 X 10-2 m/m).

RESULTS

Textural Analyses

Sand/silt/clay ratios were determined for 58 sediment samples from the Hillsboro site and are tabulated in Appendix IV. The mean values for each size fraction are:

Sand = 36.9 %

Silt = 57.3 %

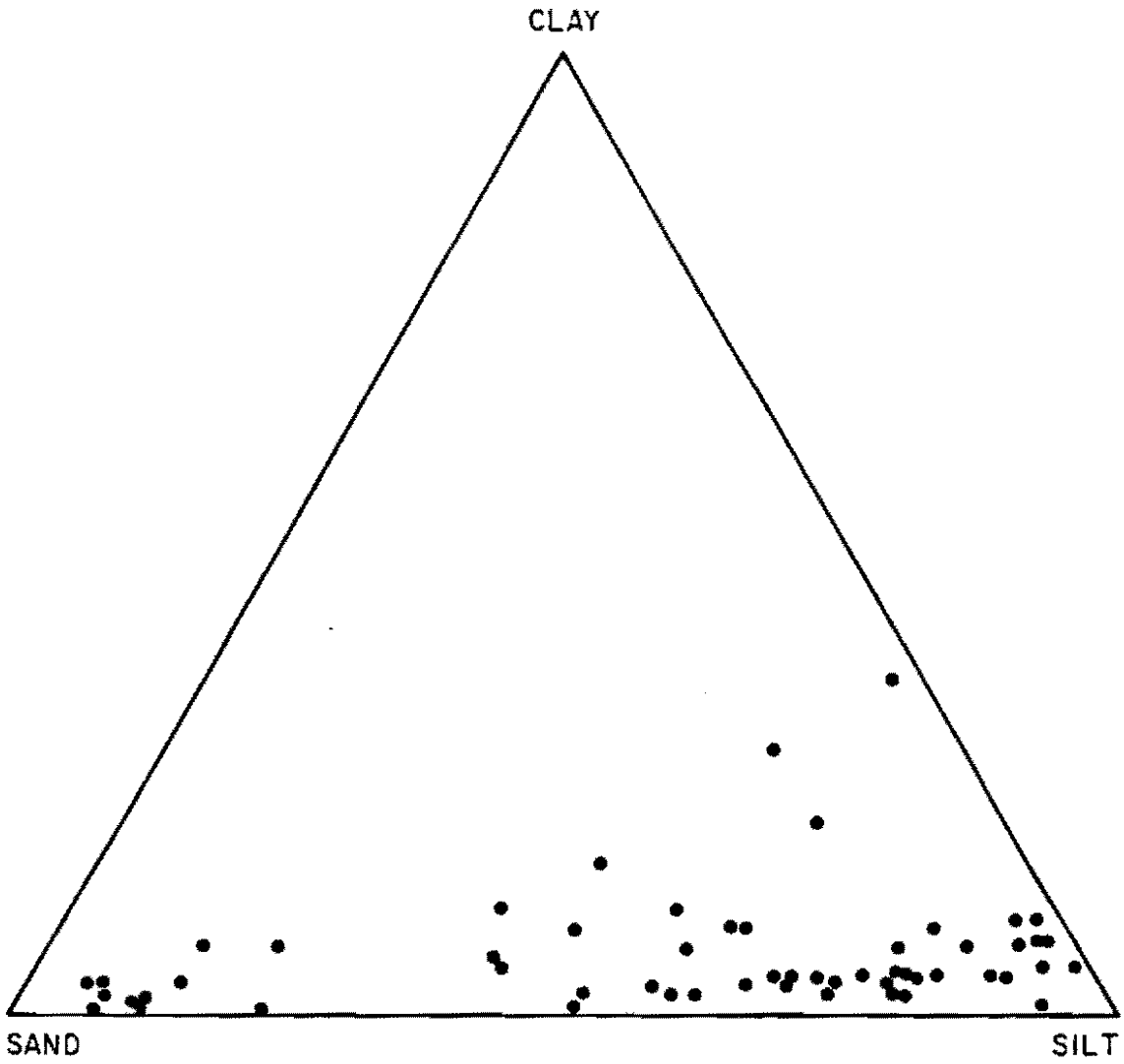
Clay = 5.8 %

Although the clay-size fraction was generally low, several samples had clay percentages greater than 20. The lowest percentage of silt was in samples from the western part of the landfill site. The highest percentages of sand occurred in samples from depths greater than 15 feet (4.5 m). The average mean grain size (d50) for the sand size fraction is 2.87 ϕ (0.137 mm), which corresponds to fine sand. The variations in the sand/silt/clay ratios are shown in a ternary diagram (Figure 14).

Water Analyses from the Saturated Zone

The results of the December, 1987 water sampling appear in Appendix V. In addition, selected chemical parameters are presented in isoconcentration maps in

Figure 14. Ternary plot of the sand/silt/clay weight ratios for the Hillsboro sediment samples from depths ranging from 2 to 80 feet (0.6 to 24.4 m) (see Appendix IV).


CLAY
SAND
SILT

Appendix VI. The results of sampling conducted in August, 1988 were unavailable for incorporation here. These results will be included in a subsequent report by the NDGS.

The chemical concentration levels were obtained from piezometers screened in silt and sand at various intervals within the zone of saturation. The background chemical concentration level was obtained from piezometer H12, approximately 1000 feet (304.8 m) northwest (upgradient) of the landfill site (Figure 7).

In general, the highest major ion concentrations were detected in piezometers H2 and H11 (Figure 12). Each of these piezometers indicated elevated total dissolved solids (TDS) concentrations and H11 indicated high chloride concentration as well. The chloride concentration in H11 was very near the recommended concentration limit (RCL) for human consumption (Table 1). The highest concentrations recorded for most parameters were centered over the covered landfill trenches; however, most concentrations did not exceed background values by more than a factor of two. The pH of the area overlying the covered trenches was slightly alkaline (7.7 - 8.0).

The highest recorded concentrations of the trace metals arsenic, cadmium, selenium, lead, copper, and chromium were significantly higher than background levels, yet well below maximum permissible concentrations (MPC) (Table 1). Both manganese and iron were detected at

TABLE 1: Recommended Concentration Limits and Maximum Permissible Concentrations for human consumption.

	mg/L
Total dissolved solids *	500
Chloride *	250
Sulfate *	250
Nitrate **	45
Iron *	0.3
Manganese *	0.05
Copper *	1.0
Zinc *	5.0
	µg/L
Arsenic **	50
Barium **	1000
Cadmium **	10
Chromium **	50
Selenium **	10
Lead **	50
Silver **	50

* Recommended Concentration Limit based upon taste and esthetic appearance.

** Maximum Permissible Concentration based upon health effects.

(after Freeze and Cherry, 1979, p. 388)

concentrations above background levels and above the MPC.

As previously indicated, water samples for organic analysis were not collected because of drought conditions. However, Ecology and Environment, Inc. did collect samples for organic analysis from piezometers H12, H13, H14, and H15 in May of 1987 (Figure 12). The parameters tested for included pthalates, acetone, toluene, chloroform, methylene chloride, 2-butanone, benzoic acid, and the PCB Arochlor 1260. The results of their analyses indicate that toluene and methylene chloride were at levels below detection limits. Acetone was detected at a higher concentration in the field decontamination blank and, therefore, not thought to be from the landfill site. Near detection limit concentrations of phthalate were detected, but were considered a plasticizer contaminant of PVC well casing. In addition, soil samples revealed low concentrations which were above background levels for 2-butanone, benzoic acid, di-n-butylphthalate, and Arochlor 1260. The small amount of Arochlor 1260 detected was attributed to agricultural activities rather than as originating from the landfill site. Thus, the results obtained by Ecology and Environment, Inc. revealed organic contaminants; however, the concentrations were thought to be insignificant (Ecology and Environment, Inc., 1987b).

X-ray Diffraction Analyses of Clays

Interpretation of the diffractograms of the samples from the Hillsboro landfill indicate that the following clay mineral phases are present: smectite, kaolinite, and muscovite/illite. The expansion of the smectite [001] reflection (as a result of ethylene glycol treatment) confirmed the presence of that mineral. The presence of kaolinite was confirmed by the 0.356-nanometer [002] reflection and the [001] reflection. The presence of muscovite/illite was confirmed using the 1.0-nanometer [001] reflection. The chloride 1.4-nanometer [001] reflection was obscured by the smectite peak. The characteristic reflections and the approximate peak locations (in degrees two theta) that were used to interpret the diffractograms are presented in Table 2. In addition to the clay minerals, several non-clay minerals were detected, including quartz and calcite. The presence of each of these minerals was also confirmed by the approximate peak locations listed in Table 2.

Peak heights, or intensities, on an x-ray diffractogram are an indication of the relative abundance of the different minerals. Accordingly the peak locations in Table 2 were used for qualitative evaluation of the Hillsboro samples. It should be noted that a number of factors contribute to difficulties in quantitative interpretation of XRD results. Overlap of weak reflections from

TABLE 2: Approximate Peak Locations in Degrees Two Theta Used in Diffractogram Interpretation

	[hkl]	2θ
Smectite (after ethylene glycol saturation)	001	5.2
Chlorite (if not obscured by smectite peak)	001	6.2
Muscovite/Illite	001	8.9
Kaolinite	001	12.4
Quartz	100	20.9
Kaolinite	002	24.9
Quartz	101	26.6
Calcite	100	29.3

(from International Centre for Diffraction Data, 1980)

other minerals with the characteristic reflections in Table 2 may obscure the presence of such minerals. Another important consideration is the fact that clay minerals display varying degrees of preferred orientation from sample to sample. This variability may result in enhancement or suppression of the characteristic reflections chosen for analysis. Perhaps the most critical factor with regard to quantitative interpretation is the difficulty in ensuring that the very small area sampled by the x-ray beam contains all minerals in proportions representative of the bulk sample. Given as normalized relative peak intensities, the proportions of the mineral phases present in the clay-size fraction are shown in Tables 3 and 4.

The results indicate that the clay-size fraction of the samples consists of a larger percentage of non-clay minerals than clay minerals. In most samples, quartz is the dominant mineral phase. Of the clay minerals present, smectite appears to be the dominant phase. However, many of the samples contain nearly equal proportions of smectite and kaolinite.

Both the relative proportion of clay minerals and the relative proportions of each of the clay types appears consistent within the samples. The relative proportions of the clay minerals is uniform for the depth intervals sampled. There does not appear to be any distinct lateral or vertical variation in the clay types for the Hillsboro

TABLE 3: Proportions of Phases Present in the Clay-size Fraction of Samples as Determined by XRD and Given as Normalized Relative Peak Intensities

Sample	Depth (ft)	Smectite [001]	Musc./Ill. [001]	Kaol. [001]	Kaol. [002]
H1B	6.5-7	60.00	26.00	_____	_____
H1C	17-22	38.32	10.08	19.75	17.69
H1D	30-32	_____	22.95	26.23	31.97
H3A	7-12	37.05	12.61	20.18	17.06
H3B	17-22	47.12	27.61	23.33	14.54
H5A	2-3	_____	43.18	34.10	_____
H5B	3-7	38.18	14.55	16.36	23.64
H5C	@7	46.15	19.23	17.31	26.92
H5D	7.5-12	33.25	10.79	19.18	15.54
H7A	2-6	25.41	13.99	11.66	10.58
H7B	7-8	25.00	7.81	10.00	13.52
H7D	22-27	61.54	13.11	25.00	26.75
H9A	2.5-5	10.46	_____	3.02	6.03
H9C	15-17.5	19.00	16.83	22.10	_____
H9D	25-27.5	25.32	12.59	17.05	16.52
H10B	5-7.5	30.66	8.37	15.09	_____
H10C	10-12	46.81	29.79	31.91	_____
H10D	12.5-15	8.79	5.41	5.68	_____
H10E	20-25	21.43	_____	5.50	_____
H11A	@17	52.09	19.67	22.87	_____

TABLE 4: Nonclay Mineral Proportions Present in the Clay-size Fraction of Samples as Determined by XRD and Given as Normalized Relative Peak Intensities

Sample	Depth (ft.)	Quartz [100]	Quartz [101]	Calcite [100]
H1B	6.5-7	_____	78.00	100
H1C	17-22	22.68	96.85	100
H1D	30-32	36.06	100	47.54
H3A	7-12	16.47	82.05	100
H3B	17-22	20.94	100	66.19
H5A	2-3	_____	100	_____
H5B	3-7	32.73	100	45.45
H5C	@7	40.38	100	53.85
H5D	7.5-12	25.37	100	27.62
H7A	2-6	9.05	53.54	100
H7B	7-8	22.15	100	74.01
H7D	22-27	16.42	100	77.32
H9A	2.5-5	_____	27.77	100
H9C	15-17.5	16.83	100	55.29
H9D	25-27.5	17.59	100	45.02
H10B	5-7.5	14.45	60.35	100
H10C	10-12	_____	100	59.57
H10D	12.5-15	47.87	100	21.63
H10E	20-25	2.78	37.35	19.21
H11A	@17	16.00	100	66.81

landfill site. Figure 15 shows the relative distribution of smectite, kaolinite, and muscovite/illite for each sample.

Hydraulic Conductivity

Hydraulic conductivity estimates for sediment within the saturated zone are presented in Table 5. The values were derived from the results of field slug tests and from textural analyses. The hydraulic conductivity values ranged from 5.41 X 10-6 to 9.18 X 10-4 ft/s (1.65 X 10-6 to 2.80 X 10-4 m/s), with an average of 2.35 X 10-4 ft/s (7.18 X 10-5 m/s). The lower values correspond to silty clay loam and the highest values to fine sand. Little difference exists between cumulative grain-size distribution curves of the samples (Appendix VII). Eight of nine hydraulic conductivity values derived from those curves are of the same order (3.28 X 10-5 ft/s (10-5 m/s)) and are representative of silt loam (Table 5). The magnitude of the values estimated for the Hillsboro site agree with those of Freeze and Cherry (1979, p. 29). Values obtained from slug tests are in general agreement with values obtained from textural analyses.

Apparent Resistivity

Apparent resistivity values for the Hillsboro landfill

Figure 15. Ternary plot of normalized relative peak intensities of smectite, kaolinite, and illite/muscovite from diffractograms of glycolated samples from the clay fraction of the Hillsboro sediment samples (see Table 3).

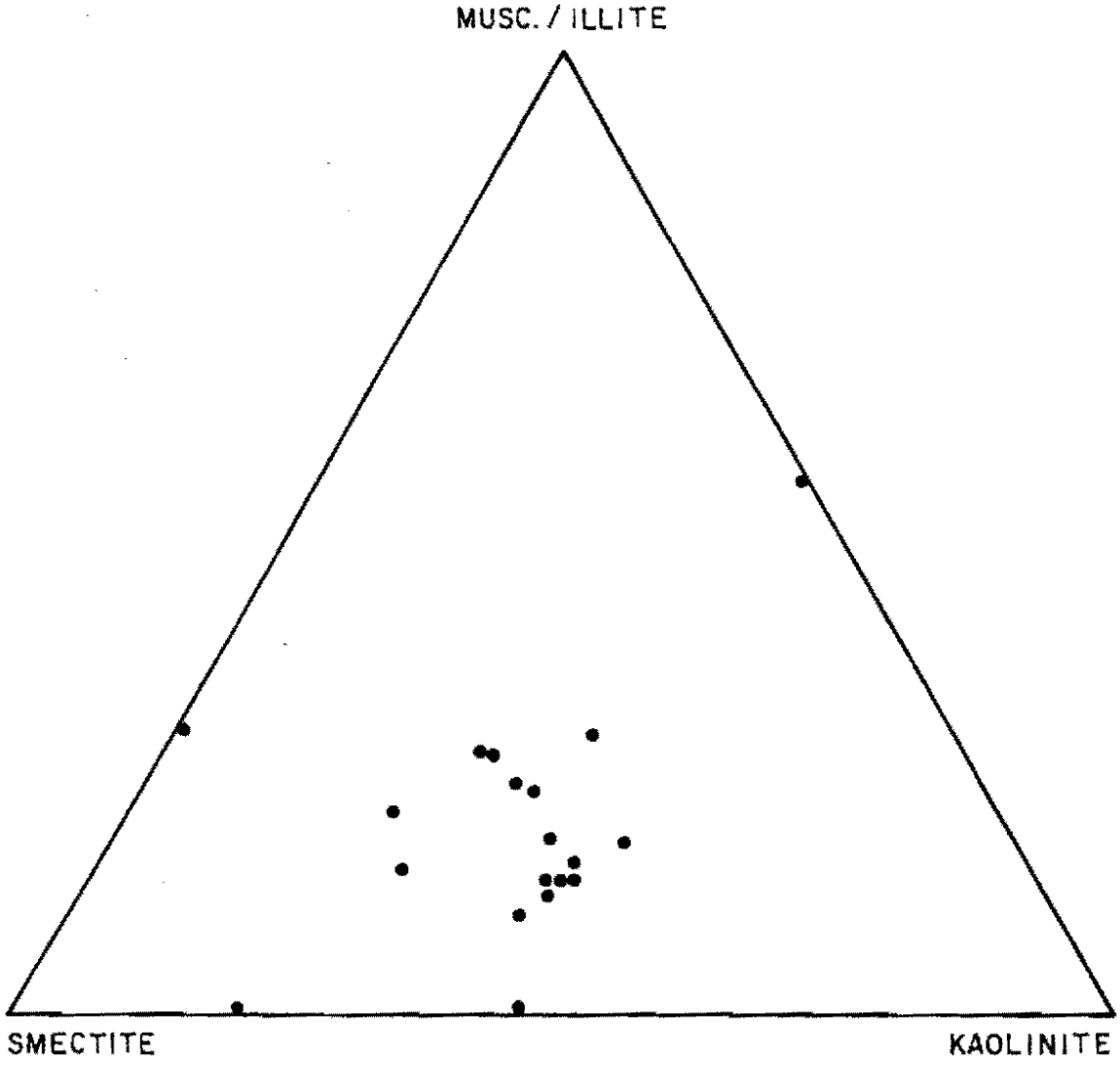

MUSC. / ILLITE
SMECTITE
KAOLINITE

TABLE 5: Hydraulic Conductivity of the Hillsboro Landfill Sediments

Piezometer	Depth (ft)	Description	Hydraulic Conductivity (m/s)
H1	30-32	Clayey silt	1.65 X 10-6
H1	2-4	Silty sand	4.95 X 10-5 *
H3	29.5-31.5	Sandy silt	1.12 X 10-5
H5	3-6	Silty sand	5.19 X 10-5 *
H5	22-27	Sand	1.01 X 10-4 *
H7	28-30	Sandy silt	1.02 X 10-5
H9	30-40	Silty sand	4.86 X 10-5 *
H9	53.5-58.5	Sand	1.33 X 10-4
H9	75-80	Sand	7.79 X 10-5 *
H10	23-25	Silty sand	2.80 X 10-4
H10	20-25	Silty sand	7.79 X 10-5 *
H11	11.5-17	Silty sand	6.10 X 10-5
H11	@17	Silty sand	4.95 X 10-5
H12	14.5-24.5	Sand	7.79 X 10-5
EW trench	16-18	Sand	7.08 X 10-5 *
Sand lenses	4-7	Sand	6.14 X 10-5 *

* Estimated from grain-size analyses, Appendix VII.

are plotted vertically in Appendix VIII. From resistivity theory it is important to remember that the resistivity measurement is affected more, proportionately, by the material at shallower depth. Accordingly, the logarithmic plot gives greater emphasis to the readings at small electrode spacings which correspond to material at shallow depths (Soiltest, Inc., 1968). Although resistivity boundaries do not necessarily correspond to stratigraphic contacts, some of the lithologies and water table depths can be correlated with the resistivity values of the log-log profiles.

Very little contrast was evident in the observed field data and high resistivity values were rare. Resistivity values generally ranged from 30 to 600 ohm-ft (9.14 to 182.87 ohm-m), throughout the study area. The apparent resistivity values are also presented in isoresistivity maps for each of the electrode spacings used at the Hillsboro site (Appendix IX). The isoresistivity maps for the electrode spacings (or depths) at or near the water table indicate a "plume" of lower apparent resistivity centered over the covered landfill trenches that extends downgradient as three lobes to the southwest, south, and southeast.

Fracture Analysis

The excavation of two trenches in the west-central portion of the landfill did not reveal an extensive network of fractures, in fact, no measurable fractures were observed and thus no qualitative results are presented here. The trenches did, however, enable close observation of the site stratigraphy. The upper 3 feet (0.9 m) of the trenches consisted of light brown clayey silt. The remainder of the exposed trench wall consisted of sandy silt with inter-bedded, fine-grained sand lenses up to 6 inches (15 cm) wide and a couple of feet (0.6 m) long (Figure 16). Numerous rootlets were also observed (Figure 17). The rootlets were approximately 1/4 inch (0.6 cm) in diameter and up to 3 feet (0.9 m) long. They were generally highly oxidized and consisted of a hollow center or a sand- or clay- filled center, and a dark outer organic layer.

Precipitation at the Hillsboro Landfill

Precipitation at the Hillsboro landfill was estimated between November of 1987 and August of 1988 by using monthly totals from the U.S. Weather Bureau gauging station at Hillsboro (Figures 18, 19, 20). During this time the Hillsboro station recorded 8.65 inches (216 mm) of precip-

Figure 16. Fine-grained sand lenses exposed in excavated trench. Tape measure shown is in inches.

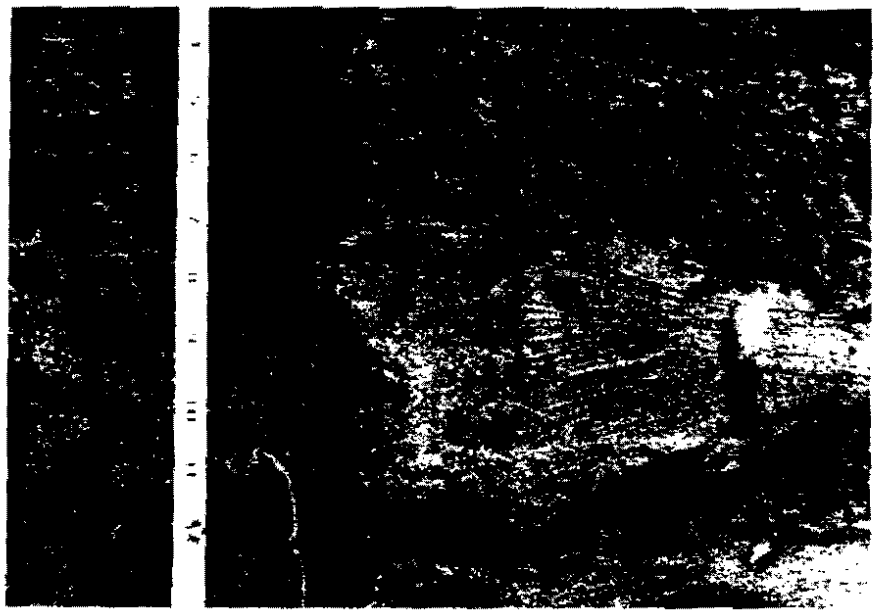


Figure 17. Rootlets observed in wall of excavated trench. Pocket knife shown for scale is approximately 2.5 inches (6.3 cm) long.

Figure 18. Water-table levels and monthly precipitation totals for the Hillsboro landfill (piezometers H1, H3, H5, and H15).

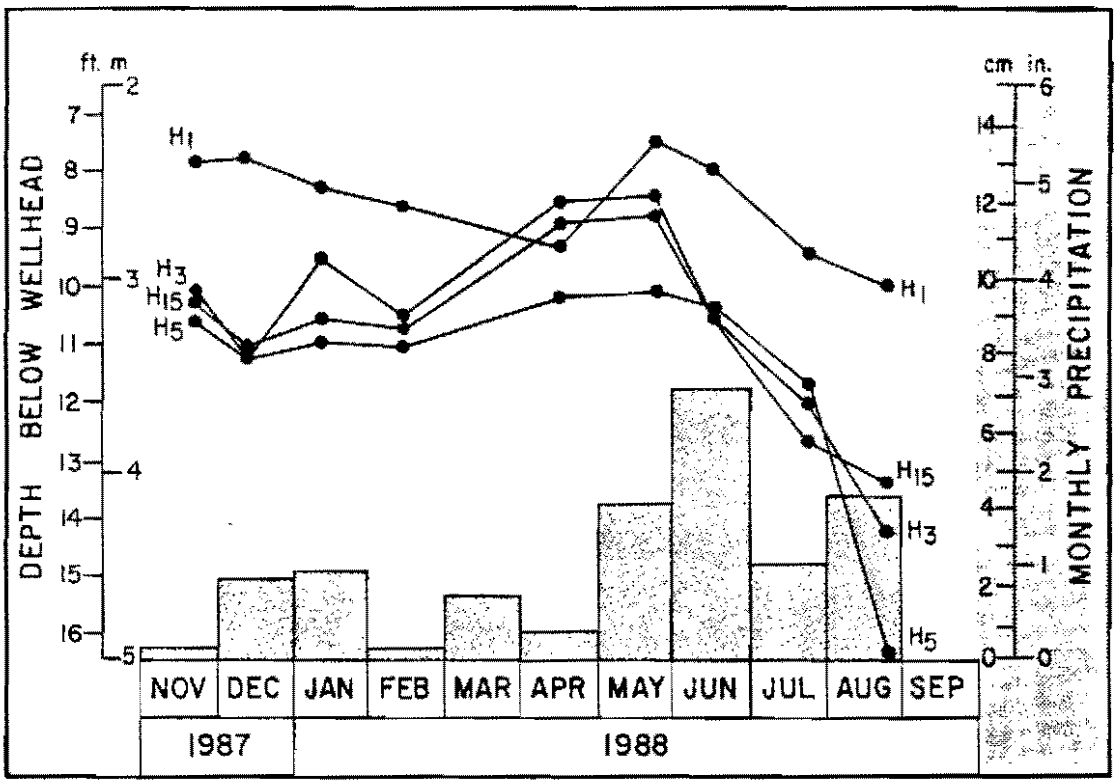

DEPTH BELOW WELLHEAD
ft. m
2
7
8
9
3
10
11
12
13
4
14
15
16
5
H1
H3
H15
H5
cm in.
6
14
5
12
10
4
8
3
6
2
4
2
1
0
0
MONTHLY PRECIPITATION
NOV
DEC
JAN
FEB
MAR
APR
MAY
JUN
JUL
AUG
SEP
1987
1988

Figure 19. Water-table levels and monthly precipitation totals for the Hillsboro landfill (piezometers H2, H4, H6, and H14).

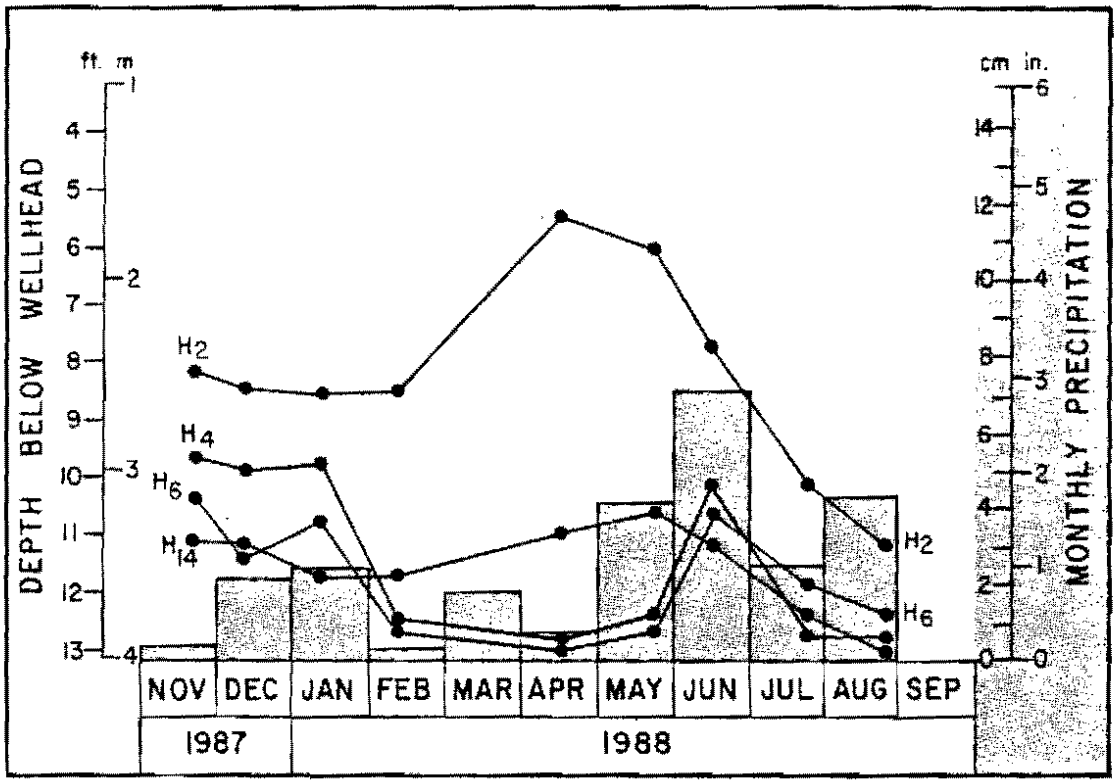
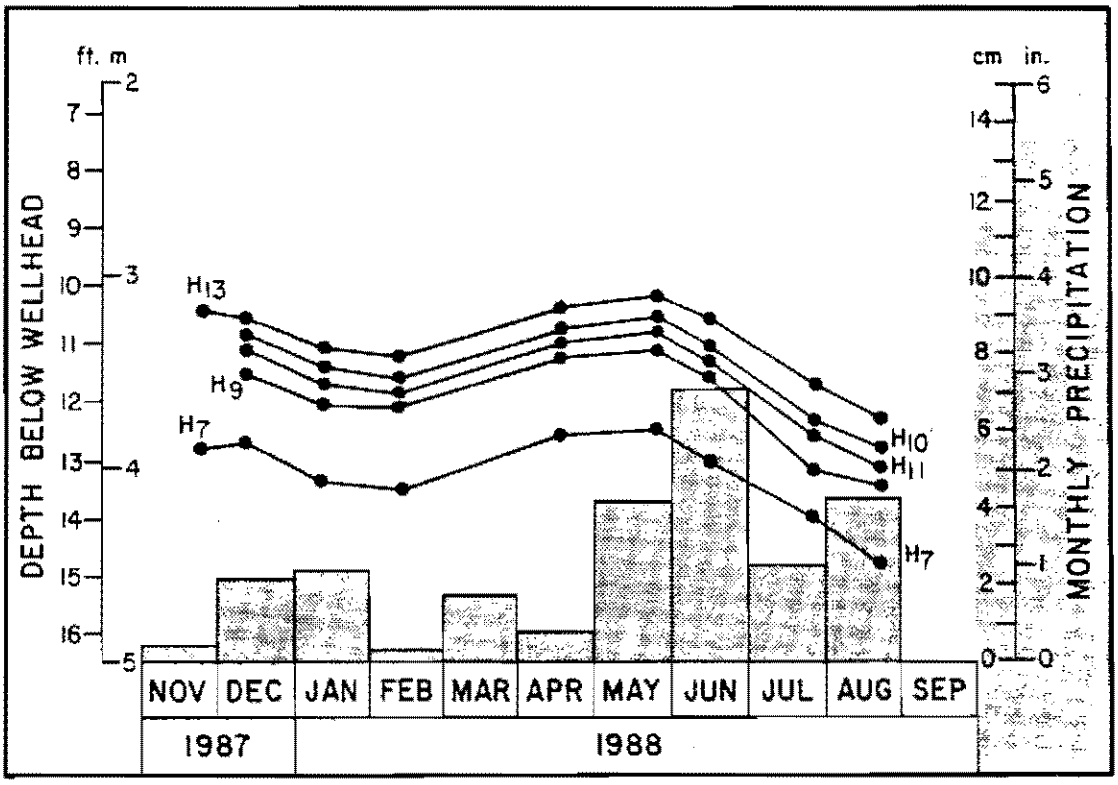

DEPTH BELOW WELLHEAD
ft. m
cm in.
MONTHLY PRECIPITATION
H2
H4
H6
H14
H2
H6
NOV
DEC
JAN
FEB
MAR
APR
MAY
JUN
JUL
AUG
SEP
1987
1988

Figure 20. Water-table levels and monthly precipitation totals for the Hillsboro landfill (piezometer H7, H9, H10, H11, and H13).


ft. m
DEPTH BELOW WELLHEAD
cm in.
MONTHLY PRECIPITATION
H13
H9
H7
H10
H11
H7
NOV
DEC
JAN
FEB
MAR
APR
MAY
JUN
JUL
AUG
SEP
1987
1988

itation. The mean precipitation for this same period (based on data from 1950-1980) is 16.30 inches (408 mm), (Ruffner, 1985). Thus, precipitation at the Hillsboro site was significantly below normal (47%).

Typically, 50 percent of the annual precipitation recorded at the Hillsboro station occurs during the months of May, June, and July (Ruffner, 1985). For 1988 this totaled only 5.5 inches (137.5 mm), more than 4.5 inches (112.5 mm) below normal. This three-month total was the fourth lowest recorded during the last 38 years (National Oceanic and Atmospheric Administration, 1988). In addition, precipitation for the month of April was nearly six times below normal. Essentially all of North Dakota experienced drought conditions during the spring and summer of 1988.

Recharge

A piezometer in which the screened interval intersects or is direcly below the water table, can be used to determine potential groundwater recharge by monitoring changes in the hydraulic head of the water table. At the Hillsboro site seven of the piezometers either intersect the water table or are screened within 3.3 feet (1.0 m) below the water table. If the water table is within 9.8 feet (3.0 m) of the ground surface, the effects of evapotranspiration and frost formation may contribute to

fluctuations of the water table, and the changes in hydraulic head may not be indicative of actual groundwater recharge (Rehm and others, 1982). Monthly water level measurements at the Hillsboro site indicate that in the northern part of the landfill, the maximum water table head was less than 9.8 feet (3.0 m) below the ground surface. Conversely, water level readings from piezometers in the southern part of the landfill indicate the maximum water table head to be greater than 9.8 feet (3.0 m) below the ground surface. Therefore, the effects of evapotranspiration and frost there are negligible.

Figures 18, 19, and 20 indicate that the water table fluctuations observed at the Hillsboro site were consistent for each of the piezometers. Water levels measured in piezometers H7, H9, H10, H11, H13, and H14 closely mirror each other (Figures 19 and 20). Measured water levels in piezometers H3 and H15 also indicate close correlation of observed water levels (Figure 18). Water levels in piezometers H4 and H6 display similar responses in that each was dry or nearly dry from February through May, and each reflects the increase in precipitation in May and June (Figure 19). In general, most of the water levels increased slightly from mid-February through late May with a subsequent steady decline throughout the summer months of June, July, and August. This reflects a water table rise coincident with the spring thaw, and a water table decline in response to prevailing drought conditions.

Recharge rates for the Hillsboro site were determined from measured hydraulic heads. The hydraulic heads were converted to vertical gradients between pairs of piezometers, which when multiplied by the measured hydraulic conductivity, yields a downward flux of water in the saturated zone (groundwater recharge rate) (Rehm and others, 1982). The resulting vertical gradient at the Hillsboro landfill averaged 1.36 X 10-1 ft/ft (4.14 X 10-2 m/m). The corresponding average vertical flux rate, or groundwater recharge rate is 2.55 X 10-6 ft/s (7.79 X 10-7 m/s).

Groundwater recharge was also calculated by multiplying the change in hydraulic head by the specific yield (Rehm and others, 1982). A specific yield value of 0.08 (Johnson, 1967) was chosen as representative of the silt loam beneath the Hillsboro site. Using this method, the groundwater recharge rate averaged 2.0 X 10-9 ft/s (8.03 X 10-10 m/s).

The difference between these values is the result of the uncertainty in the magnitude of the parameters used to calculate the recharge rates. The greatest degree of uncertainty is associated with the hydraulic conductivity and the specific yield values used in the calculation. In addition, neither of these methods takes into account the effects of evapotranspiration and frost.

Application of HELP Model

The HELP modeling program provided a simple and rapid means of simulating the amount of water movement through the Hillsboro landfill. The results of program runs based on mean monthly precipitation data and on monthly precipitation totals recorded during the period of study indicated that evapotranspiration losses were greater than or equal to precipitation totals, and, thus, no percolation occurred from the base of the landfill. Percolation was not indicated by the model until record setting precipitation totals, as was the case in 1953, were used. Even then evapotranspiration losses represented over 90 percent of precipitation totals. A peak daily precipitation total of nearly 4 inches (10 cm), as on June 15, 1953, resulted in the only measurable percolation through the landfill. The results of the HELP modeling program suggest that leachate generation from percolation occurs only during years of above normal precipitation and, even then, only during periods of intense rainfall, on the order of two to three inches (5 to 7.5 cm) in a 24-hour period.

DISCUSSION

Leachate Formation and Characteristics

The percolation of water through a landfill results in a solution with high concentrations of both organic and inorganic compounds, which is referred to as leachate. The generation of leachate is accomplished through a number of simultaneous biological, physical, and chemical changes, including biological decay of organic compounds, chemical oxidation of materials, gas evolution and diffusion, and dissolution of organic and inorganic compounds (Chen and Bowerman, 1974, p. 349). Perhaps the most important factors affecting leachate formation are the types of waste material, including organic-inorganic, degradable-nondegradable, and soluble-insoluble, and the landfill conditions, including temperature, pH, moisture, and age. Representative ranges for various inorganic constituents in leachate from sanitary landfills are shown in Table 6.

The refuse for sanitary landfills comes primarily from ordinary household and commercial solid waste. Typically, paper comprises the largest share of sanitary landfill refuse, approximately 37 percent (Environmental Protection Agency, 1986b). Paper is usually quite low in moisture content, however, and increased amounts of paper in refuse have resulted in decreased rates of refuse breakdown (Chen and Bowerman, 1974, p. 51). Conversely, food wastes

TABLE 6: Concentration Ranges of Inorganic Leachate Constituents in Municipal Landfills.

Constituent	Concentration Range
Alkalinity	0.1-20,350 mg/L
Cadmium	0-0.375 mg/L
Chloride	30-5,000 mg/L
Hardness	0.1-36,000 mg/L
Lead	0.001-1.44 mg/L
Nitrate	0.1-45 mg/L
pH	3.5-8.5
Selenium	0-2.7 mg/L
Sulfate	25-500 mg/L
Total Dissolved Solids	725-55,000 mg/L
Turbidity	30-450
Conductivity	960-16,300 micromhos/cm
Copper	0.1-9 mg/L
Iron	200-5,500 mg/L
Magnesium	3-15,600 mg/L
Manganese	0.6-41 mg/L
Potassium	35-2,300 mg/L
Sodium	20-7,600 mg/L
Chromium	0.02-18 mg/L
Zinc	0.6-220 mg/L

(after Environmental Protection Agency, 1986b, p. 4-12)

comprise approximately 8 percent of sanitary refuse and may be extremely wet. In general, the moisture content of mixed solid waste generated by a community ranges from 20 to 30 percent by weight (Weiss, 1974, p. 11). Moisture content alone should not produce leachate.

Migration and Attenuation of Leachate

The movement of landfill leachate in groundwater is governed by the interaction of advection, dispersion, and molecular diffusion. Advection refers to the transport of contaminants at the same speed as the average linear velocity of groundwater. In most cases this is true of the nonreactive solutes, generally the dissolved anions (Freeze and Cherry, 1979, p. 75). Dispersion involves the spreading of a volume of contaminants as it flows through the subsurface. This spreading initiates mixing with uncontaminated groundwater and ultimately results in dilution of the contaminant plume. Dispersion may result in leachate transport at velocities greater than the average groundwater velocity. Molecular diffusion is commonly included as a component of microscopic dispersion (Anderson, 1984). Molecular diffusion occurs as ions or molecules move from sites of higher to lower concentrations.

Leachate movement may be slowed or stopped by chemical attenuation of inorganic contaminants involving adsorption, precipitation, and oxidation or reduction. Foremost of

these controls is adsorption by clay minerals. Griffin and others (1976) found that the attenuating ability of clays was directly related to their cation exchange capacities. They further noted that the most highly attenuated ions are the trace metals.

Chemical Indicators of Contamination

Constituents most helpful in distinguishing groundwater contaminated by landfill leachate include specific conductance, ammonium, chloride, bicarbonate, iron, potassium, and sulfate (Saar and Braids, 1983). The usefulness of these indicators generally reflects their low susceptibility to attenuation and higher concentrations within landfill leachate relative to natural occurrence.

Leachate Migration at the Hillsboro Landfill

Although contaminant levels appear to be low, the chemical analyses (Appendix V) conducted on water samples from the Hillsboro landfill suggest that degradation of the groundwater beneath the landfill has occurred. The isoconcentration maps (Appendix VI) reveal an increase in ion concentrations over the background levels, most notably the trace metals arsenic, cadmium, selenium, lead, copper, chromium, iron, and manganese. The concentrations of these trace metals ranged from 3.5 to 20 times above background

levels. The isoconcentration maps indicate a contaminant plume centered over the covered landfill trenches, extending downgradient as lobes to the southwest, south, and southeast. Reduced ion concentrations were observed downgradient, away from the buried refuse. Major cations such as potassium, sodium, magnesium, and calcium are assumed to undergo concentration changes primarily due to ion exchange. The trace metals may be limited in occurrence due to solid phase solubility controls involving sulfide, hydroxide, and carbonate (Cherry, 1983).

The configuration of the plumes on the isoconcentration maps is believed to be the result of longitudinal and transverse dispersion within the saturated zone. The shape and position of the plumes indicate that the buried refuse is the source of contamination. However, it is apparent that plume shape is not consistent for all the parameters. The burial history of the refuse at the Hillsboro landfill is not well documented. Thus, it is believed that the plume shapes for some parameters reflect heterogeneities of the refuse within the landfill trenches and isolated disposal or surface spills outside of the covered trenches. For example, arsenic appears in many pesticides. The burial of empty pesticide containers in the northwest corner of the landfill may have contributed to the configuration of the arsenic plume. Isolated burial or surface spills of lead acetate (accepted from the American Crystal Sugar Company) may account for the lead plume. The

plume configuration for a number of parameters may have been influenced by burn piles throughout the years. Typically such burn piles contain large amounts of construction debris, including gypsum wall board, insulation materials, and roofing materials.

The higher levels of iron and manganese observed at the Hillsboro site may have been derived from the sand beneath the landfill; the redox conditions beneath landfills can cause dissolution of oxide coatings of iron and manganese on sand grains (Cherry, 1983). However, iron may have come from stockpiling of iron which occurred on site during the fall of 1987 and possibly in the past as well. These two elements are apparently less influenced by adsorption.

Factors perceived as contributing to the apparent low levels of released contaminants include attenuation by clays, low levels of precipitation, coupled with high evapotranspiration losses, and the relatively short residence time of the buried refuse. With regard to the attenuation capabilities of the clays at the Hillsboro landfill it should be reiterated that the clay-size fraction of the samples is small (Appendix IV), and the proportion of non-clay minerals within the clay-size fraction is large (Table 4). However, Griffin and others (1976) noted that sediment columns containing only two percent smectite effectively attenuated landfill leachate constituents, most notably the heavy metals. Griffin and

others (1976) attributed the attenuating ability of clays to their cation exchange capacities. Of special importance in smectite is the sodium-calcium exchange reaction (Freeze and Cherry, 1979, p. 133). Replacement of calcium by sodium results in expansion of the smectite clay structure which can effectively decrease permeability. It is expected that clay adsorption and ion exchange within the clayey silt and sandy silt intervals, which contain at least 10 percent clay-size sediment, are important mechanisms limiting the mobility of contaminants.

The effects of evapotranspiration were surely more pronounced in light of the drought conditions experienced during the period of study. Measured evapotranspiration in North Dakota is on the order of 1.5 times greater than precipitation (Rehm and others, 1982, p.18), a scenario substantiated by the HELP computer model. In spite of this, recharge can occur, dependent on the intensity and duration of precipitation. Significant recharge in the summer months likely occurs only when very high intensity rains result in high enough infiltration rates for the water to quickly pass through the root zone. During the summer of 1988, such recharge events were virtually nonexistent. The lack of infiltration and subsequent recharge suggests that very little if any leachate should reach the water table at the Hillsboro site. However, because the depth to the water table is between 5.4 and 11 feet (1.66 and 3.37 m) below the surface in the northern

part of the landfill, burial of refuse in trenches 15-feet (4.57 m) deep ensure that the lower portion of the landfill would become saturated. It is expected that the trenches were designed with a slight northward slope, and as such, these saturated conditions within the refuse were more prevalent in the northern part of the landfill. Thus, even without infiltration through the landfill cover, leachate would be expected to be produced through lateral migration of groundwater into the refuse-filled trenches.

Another factor influencing the production of leachate is time. The Hillsboro landfill was in operation less than 12 years. Given the likelihood that the buried refuse consists of large amounts of paper and that large portions of the landfill have remained relatively dry, it appears as though 12 years should not be considered an unusually long period of time with regard to leachate generation. In fact, given the calculated average linear velocity of 8.59 X 10-7 ft/s (2.62 X 10-7 m/s), groundwater travel time from piezometer H1 to H14 is nearly 28 years. Similar investigations in other landfills throughout North Dakota suggest that it is not uncommon to find newspaper that has been buried in excess of ten years and yet displays little evidence of breakdown (Tillotson, 1988). At the Hillsboro landfill, a shallow test hole in the oldest portion of the site revealed much the same (Figure 21). Steiner and others (1971) note that leachate appearance may be offset from the initial time of emplacement by as much as 20

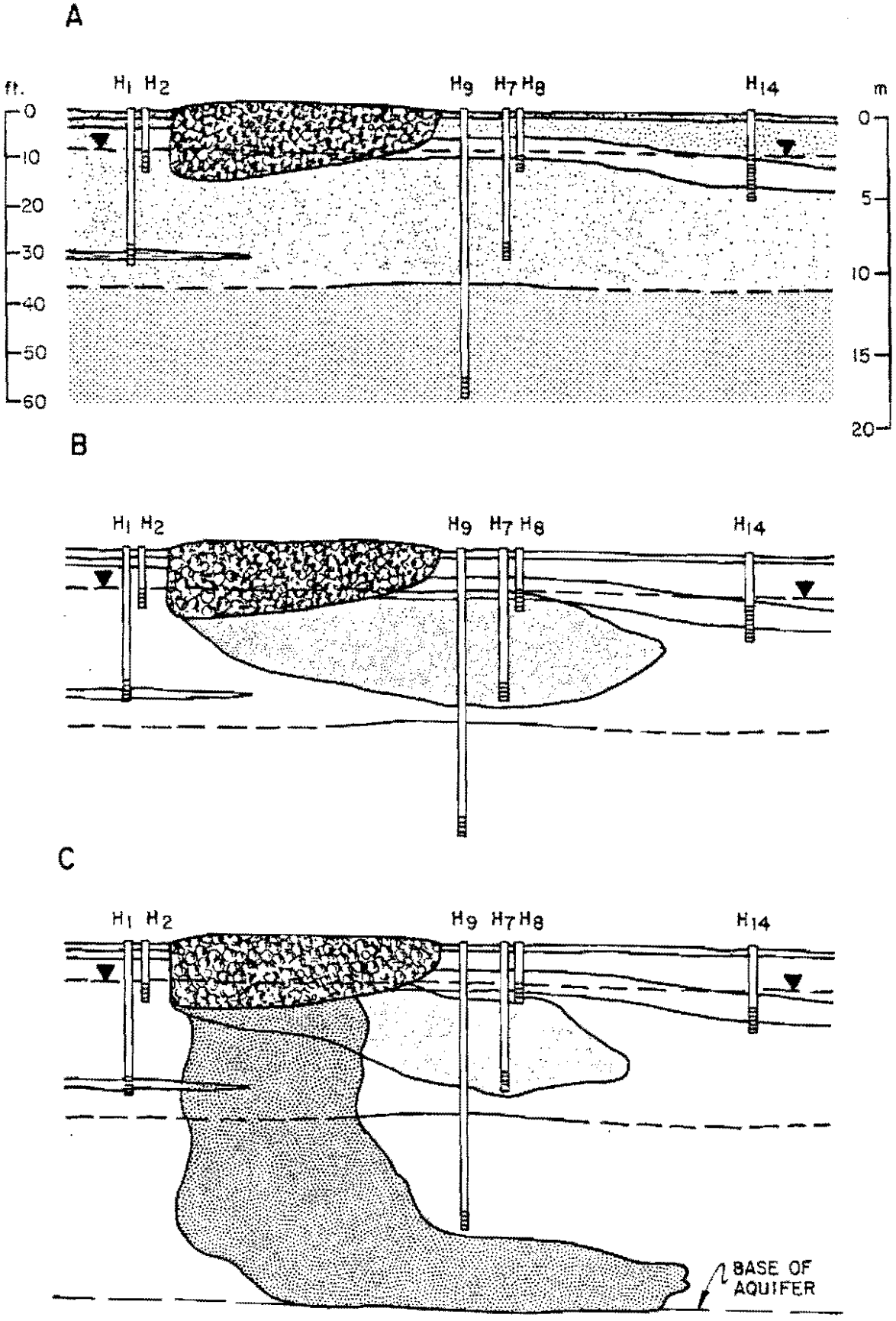
Figure 21. Newspaper recovered from a shallow test hole in the oldest portion of the landfill site.

years, and that short-term studies may be inadequate to establish the magnitude of the problem.

In addition to the preceding discussion, a number of variables, including placement of the piezometers, depths of screened intervals, and the parameters tested for, may account for the apparent low levels of contaminants. The parameters tested for did reveal a pattern; however, several other parameters useful in leachate plume delineation, including ammonium, boron, mercury, and freon were not analyzed for. Also, organic analyses, including total organic carbon (TOC) may have revealed more extensive groundwater contamination beneath the Hillsboro site.

Two possible alternatives of contaminant movement at the Hillsboro landfill are presented in Figure 22. In the first case, low levels of contaminants within the leachate plume are detected downgradient of the buried refuse (Figure 22B). In the second case (Figure 22C), due to the relatively high vertical gradient beneath the landfill site, the leachate plume moves primarily downward and remains directly under the contamination source, and subsequent movement with the groundwater flow allows the main body of the plume to pass undetected below the monitoring equipment. However, it appears unlikely that such plume movement would not be detected by piezometer H9. The possible existence of a deep plume was the reason for attempting to install piezometer H9 at a depth of 82 feet (25 m).

Figure 22. Two alternative contaminant plume configurations beneath the Hillsboro landfill. (A) North-south cross-section at the landfill displaying the location of piezometers, the position of the water table, and the geology beneath the site. (B) Low levels of contaminants are detected downgradient of the buried refuse and/or (C), because of a relatively high vertical gradient, the leachate plume moves primarily downward and remains directly under the contamination source. For lithologic symbols see Figure 13.


A
ft.
0
10
20
30
40
50
60
H1 H2
H9 H7 H8
H14
m
0
5
10
15
20
B
H1 H2
H9 H7 H8
H14
C
H1 H2
H9 H7 H8
H14
BASE OF AQUIFER

Apparent Resistivity

Quantification of surface resistivity results for comparison with the results of water sampling was attempted for the Hillsboro site. If the apparent resistivity values reflect the quality of groundwater, then isoresistivity contours (Appendix IX) should parallel isoconcentration contours (Appendix VI) (Klefstad and others, 1975).

The contours of the isoconcentration maps for potassium and chloride roughly parallel the isoresistivity maps for the 8-, 10-, 12-, and 16-foot (2.4-, 3.1-, 3.7-, and 4.9-metre) electrode spacings. The contours of the TDS and bicarbonate isoconcentration maps correlate to a lesser degree with the isoresistivity maps. The high TDS concentrations in H2 and H11 do not appear to be reflected in the field data. In general, the change in resistivity sought as an indicator of contamination was small and essentially undetectable. A slight reduction in resistivity values centered over the covered trenches may be attributed to the lack of vegetative cover at the time of the survey (Murphy and Kehew, 1984).

With regard to correlation between apparent resistivity and lithology, perhaps the best results were obtained at resistivity station 3 (Appendix VIII). The presumably low moisture content of the near-surface sandy silt account for the highest resistivity values. The

influence of a clayey layer at approximately 24 feet (7.3 m) is reflected in the lowest resistivity values, whereas the cleaner sand at depth again produces higher values. In general, the log-log plots of apparent resistivity versus electrode spacing reflect the higher near-surface values of dry sand and sandy silt, the lower uniform values of saturated sandy silt, the lowest values of clayey layers, and the higher values of a clean sand at depth. In some cases, the high silt content and the low degree of consolidation of these deposits may have attributed to the poor definition of layer boundaries (Murphy and Kehew, 1984).

Interpreted Resistivity

The apparent resistivity values obtained reflect the true resistivity of the geologic sediments only if they are homogeneous and isotropic (Yazicigil and Sendlein, 1982). As this is rarely the case in the subsurface, a computer program developed by Zohdy and Bisdorf (1988) was used to interpret the apparent resistivity values and automatically calculate layer thicknesses and resistivities (Appendix VIII). Profiles of interpreted resistivity values for the Hillsboro site are in general agreement with the log-log plots of apparent resistivity versus electrode spacing. The position of the water table appears to be accurately represented by the interpreted profiles.

A major shortcoming of the computer program rests in modifications made to the PC version to enable the interpretation of Wenner sounding data. These modifications may have introduced as much as 10 percent error into the data interpretation (Bisdorf, 1988). In addition, Kehew and Groenewold (1983) have pointed out that at the larger electrode spacings, very small changes in potential difference result in significant differences in apparent resistivity. Yet the sensitivity of the computer program is such that small changes in the slope of the apparent resistivity curve can profoundly alter the interpreted layering sequence. Thus, in sediments with a high degree of electrical uniformity, correlation between sediment boundaries and the computer-generated sequence of layers is difficult.

Limitations in Earth Resistivity Surveying

Successful application of resistivity methods to groundwater contamination investigations depends on favorable conditions, including uniform subsurface conditions, a shallow groundwater table, and a significant conductivity contrast between contaminated and natural groundwater (Klefstad and others, 1975). If these conditions are not met, they impose significant limitations which can preclude the success of the resistivity survey.

A number of conditions favorable to resistivity

surveying exist at the Hillsboro landfill, including a shallow groundwater table and relatively uniform subsurface conditions. However, very little contrast was evident in the observed field data. The interpretation and correlation of resistivity data are more accurate when orders of magnitude contrasts occur (Kehew and Groenewold, 1983).

CONCLUSIONS

The Hillsboro landfill is representative of a landfill that was located, operated, and designed with little or no consideration of the geologic and hydrogeologic conditions of the site. Its location above a major aquifer, within permeable surface materials, and within shallow water table conditions should not have been permitted. Operation as an open dump and acceptance of lead acetate from the American Crystal Sugar Company, coupled with the fact that the landfill cells are unlined and at least portions of the buried refuse are below the water table, has undoubtedly increased the potential for groundwater contamination beneath this site.

Although the water sampling activities conducted as part of this project were limited by financial and time constraints, as well as by drought conditions, the results of this study do, indeed, reveal that groundwater degradation beneath the buried refuse has occurred. In addition, the following can be concluded:

1. The depth to the water table varied from 5.4 to 11 feet (1.7 to 3.4 m) below the surface in the northern part of the landfill and from 10.6 to 13.2 feet (3.2 to 4.0 m) below the surface in the southern part of the landfill site.

2. Water table maps indicate that groundwater beneath the site is flowing to the south-southeast, away from covered landfill trenches.

3. The silty clay loam, silty loam, silt, and sand, which define the stratigraphy of the Hillsboro landfill, are characteristic of deposition in proglacial lakes and the fluvial influence of drainage into these shallow lakes.

4. Smectite is the dominant clay mineral present in the Hillsboro sediments.

5. The amount of leachate generated at the Hillsboro landfill is small because of: a) normally low amounts of infiltration, b) absorption of water by large amounts of paper within the compacted buried refuse, c) comparatively little decay as a result of the relatively short residence time of the buried refuse, d) attenuation by smectite clays, and e) dilution by mixing with uncon-taminated groundwater.

6. Although well below maximum permissible concentrations, the concentrations of the trace metals arsenic, cadmium, selenium, lead, copper, chromium, iron, and manganese ranged from 3.5 to 20 times more than background levels.

7. Quantification of electrical earth resistivity results

(for comparison with the results of water sampling) proved difficult due to the low concentration of contaminants and the associated negligible contrast between the conductivities of contaminated and natural groundwater.

8. In spite of the low levels of contaminants detected, the position of the shallow water table within the lower portions of the refuse-filled trenches poses a significant threat to groundwater beneath the landfill.

RECOMMENDATIONS

Implicit in the description of the Hillsboro landfill is the recommendation to avoid such locations for waste disposal. The inadequacies of the site and the potential for groundwater contamination beneath the site ultimately resulted in the closure of the landfill and the initiation of this study. Thus, given the potential for groundwater contamination as suggested by the setting of the landfill and by the results of this study, groundwater monitoring should continue at this site. Also, given the limitations imposed on water sampling by drought conditions, such monitoring should include a wet spring during which normal recharge occurs and optimum leachate production and movement might be expected.

Although high levels of contaminants were not detected, it is expected that organic sampling may provide useful indicaters of leachate production. Additional water sampling, including organic analysis, is appropriate.

The cation exchange capacities of the Hillsboro sediments should be determined. This is important in light of the fact that the attenuating ability of clays is a function of their cation exchange capacities.

Concern over the proximity of the landfill to the Hillsboro aquifer warrants more accurate definition of the aquifer boundaries beneath the site. Accordingly, several deep (50 to 100 feet (15.2 to 30.5 m)) test holes should be

drilled. Such drilling should include the installation of at least one deep piezometer (> 80 ft (24.4 m)) in order to better evaluate the characteristics of groundwater within the aquifer.

The placement of additional piezometers within the present monitoring scheme should also be considered. Installation of a piezometer nest near H12 (Figure 7), including the addition of both a shallower and deeper piezometer, would permit better interpretation of background conditions. Other possible additions might include piezometer nests incorporating the downgradient wells H13, H14, and H15 (Figure 7). Due to the shallow water table at the site, instrumentation of the vadose zone is probably unnecessary. The incorporation of these suggestions would serve to strengthen aspects of this study which were limited by financial and time constraints, and by drought conditions.

APPENDICES

APPENDIX I

Piezometer Elevations and Screened Intervals

Piezometer	Surface Elevation (ft)	Screened Interval (ft below the surface)
H1	930	30-32
H2	930	7-12
H3	931	29.5-31.5
H4	931	7.5-12.5
H5	932	28-30
H6	932	7-12
H7	935	28-30
H8	935	7-12
H9	932	53.5-58.5
H10	932	23-25
H11	932	11.5-17
H12	933	14.5-24.5
H13	932	11.9-21.9
H14	931	8-18
H15	930	7.5-17.5

APPENDIX II

Lithologic Description of Drill Holes

H1

Depth (ft)	Description
0-2	Fill
2-4	Sand, dark grayish brown (10YR,4/2), color wet=very dark grayish brown (10YR, 3/2), fine-medium grained, moderately sorted, subangular-subrounded(0.5) sphericity(0.7).
4-17	Sand and silt, light yellowish brown (10YR, 6/4), color wet= brown (10YR, 4/3), fine-medium grained, subangular-subrounded, some angular grains (0.3), sphericity (0.5-0.7), FeO stain on sand grains.
17-22	Sand and silt, very pale brown (10YR, 7/4), color wet=dark yellowish brown (10YR, 4/4), fine-grained, subrounded (0.5), well sorted, sphericity (0.3).
22-27	Sand and silt, light yellowish brown (10YR, 6/4), color wet= brown (10YR, 4/3), fine-grained, well-sorted, angular-subrounded (0.3), sphericity (0.5).
27-30	Sand and silt, light brownish gray (10YR, 6/2), color wet=grayish brown (10YR, 5/2), fine-medium grained, well sorted, subangular-subrounded (0.5), sphericity (0.7).
30-32	Sand, silt, and some clay, gray (10YR, 6/1), color wet=dark gray (10YR, 4/1), v. fine-fine grained, well sorted.

H2

Depth (ft)	Description
0-2	Fill
2-4	Sand, dark grayish brown (10YR,4/2), color wet=very dark grayish brown (10YR, 3/2), fine-medium grained, moderately sorted, subangular-subrounded(0.5) sphericity(0.7).
4-12	Sand and silt, light yellowish brown (10YR, 6/4), color wet= brown (10YR, 4/3), fine-medium grained, subangular-subrounded, some angular grains (0.3), sphericity (0.5-0.7), FeO stain on sand grains.

H3

Depth (ft)	Description
0-2	Topsoil
2-5	Sand and silt, medium brown, fine to medium grained.
5-7	Sand and silt, light yellow brown, fine-medium grained.
7-9	Sand and silt, alternating red (2.5YR, 5/6) and gray (10YR, 6/1) layers, very fine-grained, well sorted, subrounded.
9-22	Sand and silt, light yellowish brown (10YR, 4/6), color wet= dark yellowish brown (10YR, 4/6), v. fine-fine grained, well

	sorted, subrounded, sphericity (0.7).
22-32	Sand and silt, pale brown (10YR, 6/3), color wet=dark grayish brown (10YR, 4/2), similar to above.

H4

Depth (ft)	Description
0-2	Topsoil
2-5	Sand and silt, medium brown, fine to medium grained.
5-7	Sand and silt, light yellow brown, fine-medium grained.
7-9	Sand and silt, alternating red (2.5YR, 5/6) and gray (10YR, 6/1) layers, very fine-grained, well sorted, subrounded.
9-12.5	Sand and silt, light yellowish brown (10YR, 4/6), color wet= dark yellowish brown (10YR, 4/6), v. fine-fine grained, well sorted, subrounded, sphericity (0.7).

H5

Depth (ft)	Description
0-1	Topsoil
1-3	Sand, brown (10YR, 5/3), color wet=dark brown (10YR, 3/3), fine-medium grained, moderately sorted, subangular (0.3), sphericity (0.9).

3-6	Sand and silt, very pale brown (10YR, 7/3), color wet=dark yellowish brown (10YR, 4/4), very fine-fine grained, well sorted, subrounded (0.5), sphericity (0.7), FeO stained.
6-7.5	Clay and silt, very pale brown (10YR, 7/3), and yellowish red (5YR, 5/8), color wet=yellowish brown (10YR, 5/4) and dark yellowish brown (10YR, 4/4), FeO stained.
7.5-12	Sand and silt, light yellowish brown (10YR, 6/4), color wet=dark yellowish brown (10YR, 4/4), very fine grained, moderately sorted, subangular (0.3), sphericity (0.9).
12-22	Sand, some silt, pale brown (10YR, 6/3), color wet=dark brown (10YR, 3/3), medium-coarse grained, well sorted, subangular to subrounded (0.5), sphericity (0.7-0.9).
22-32	Sand, pale brown (10YR, 6/3), color wet=dark brown (10YR, 3/3), fine to medium grained, very well sorted, subrounded to rounded (0.5), sphericity (0.9).

H6

Depth (ft)	Description
0-1	Topsoil
1-3	Sand, brown (10YR, 5/3), color wet=dark brown (10YR, 3/3), fine-medium grained, moderately sorted, subangular (0.3), sphericity (0.9).
3-6	Sand and silt, very pale brown (10YR, 7/3), color wet=dark yellowish brown (10YR, 4/4), very fine-fine grained, well sorted,

	subrounded (0.5), sphericity (0.7), FeO stained.
6-7.5	Clay and silt, very pale brown (10YR, 7/3), and yellowish red (5YR, 5/8), color wet=yellowish brown (10YR, 5/4) and dark yellowish brown (10YR, 4/4), FeO stained.
7.5-12.5	Sand and silt, light yellowish brown (10YR, 6/4), color wet=dark yellowish brown (10YR, 4/4), very fine grained, moderately sorted, subangular (0.3), sphericity (0.9).

H7

Depth (ft)	Description
0-2	Fill
2-6	Sand, light yellowish brown (10YR, 6/4), color wet=dark yellowish brown (10YR, 4/4), very fine grained, well sorted, subrounded (0.5), sphericity (0.7), FeO stained, micaceous.
6-12	Silt and clay, very pale brown (10YR, 7/3) and yellowish red (5YR, 5/8), color wet=dark brown (10YR, 4/3), FeO stained.
12-17	Sand and silt, light yellowish brown (10YR,4/4), color wet=dark brown (10YR, 4/3), very fine grained, well sorted, subrounded (0.5), sphericity (0.9), FeO stained, some mica.
17-32	Sand, some silt, light yellowish brown (10YR, 4/4), color wet=dark brown (10YR, 4/3), very fine grained, well sorted, subrounded (0.5), sphericity (0.9), FeO stained, micaceous.

H8

Depth (ft)	Description
0-2	Fill
2-6	Sand, light yellowish brown (10YR, 6/4), color wet=dark yellowish brown (10YR, 4/4), very fine grained, well sorted, sub-rounded (0.5), sphericity (0.7), FeO stained, micaceous.
6-12	Silt and clay, very pale brown (10YR, 7/3) and yellowish red (5YR, 5/8), color wet=dark brown (10YR, 4/3), FeO stained.

H9

Depth (ft)	Description
0-2	Fill
2-5	Sand and silt, very pale brown (10YR, 7/3), color wet=yellowish brown (10YR,5/4), fine grained, well sorted, subangular to sub-rounded (0.5), sphericity (0.7), some FeO staining.
5-7.5	Sand, pale brown (10YR, 6/3), thinly laminated, color wet= dark yellowish brown (10YR, 4/4), fine-grained, well sorted, sub-angular to subrounded (0.5), sphericity (0.7).
7.5-15	Sand and silt, light gray (10YR, 7/1) and light yellowish brown (10YR, 6/4), color wet=grayish brown (10YR, 5/2) and yellowish brown (10YR, 5/6), very fine grained, well sorted, subrounded, FeO stained, thinly laminated in

	lower 5 feet.
15-24	Sand and silt, pale brown (10YR, 6/3), color wet=dark brown (10YR, 4/3), very fine to fine grained, well sorted, subrounded (0.5), sphericity (0.7).
24-25	Clay and silt, gray.
25-30	Sand and silt, pale brown (10YR, 6/3), color wet=dark grayish brown (10YR, 4/2), very fine grained, micaceous.
30-40	Sand, light brownish gray (10YR, 6/2), color wet=grayish brown (10YR, 5/2), medium grained, subangular to subrounded (0.3-0.5), sphericity (0.9), moderately sorted.
40-82.5	Sand, gray (10YR, 6/1), color wet=gray (10YR, 4/1), salt and pepper, medium to coarse grained, moderately sorted, subangular to rounded (0.3-0.7), sphericity (0.7-0.9), some FeO stain, sand coarsens downward.

H10

Depth (ft)	Description
0-2	Fill/Topsoil
2-2.5	Clay, light gray (10YR, 7/1), color wet=brown (10YR, 4/3).
2.5-5	Sand, light yellowish brown (10YR, 6/4), color wet=yellowish brown (10YR, 5/4), very fine grained, well sorted, subangular (0.3), sphericity (0.7), FeO stained, some mica flakes.
5-12.5	Sand, pale brown (10YR, 6/3), color wet=yellowish brown (10YR, 5/4), fine grained, moderately

	sorted, subangular-subrounded (0.3-.5), sphericity (0.7), thinly laminated, FeO stained lamina prominant, micaceous.
12.5-17	Sand, pale brown (10YR, 6/3), color wet=dark brown (10YR, 4/3), laminae absent, fine-medium grained, subrounded-well rounded (0.7), moderately sorted, sphericity (0.7-0.9).
17-17.5	Clay and silt, very pale brown (10YR, 7/4), coloR wet=yellowish brown (10YR, 5/4).
17.5-25	Sand and silt, pale brown (10YR, 6/3), color wet=dark brown (10YR, 3/3), medium grained, moderately sorted, subrounded to rounded (0.7), sphericity (0.9).

H11

Depth (ft)	Description
0-2	Fill/Topsoil
2-2.5	Clay, light gray (10YR, 7/1), color wet=brown (10YR, 4/3).
2.5-5	Sand, light yellowish brown (10YR, 6/4), color wet=yellowish brown (10YR, 5/4), very fine grained, well sorted, subangular (0.3), sphericity (0.7), FeO stained, some mica flakes.
5-12.5	Sand, pale brown (10YR, 6/3), color wet=yellowish brown (10YR, 5/4), fine grained, moderately sorted, subangular-subrounded (0.3-0.5), sphericity (0.7), thinly laminated, FeO stained lamina prominant, micaceous.
12.5-17	Sand, some silt, light yellowish brown (10YR, 6/4), color wet= dark yellowish brown (10YR, 3/4),

very fine to fine-grained, well sorted, subrounded to rounded (0.7), sphericity (0.9), FeO stained.

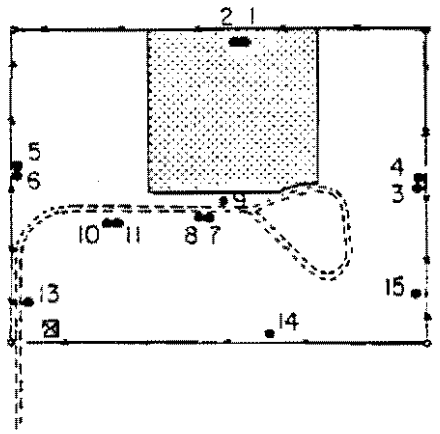
APPENDIX III

Water Table Maps

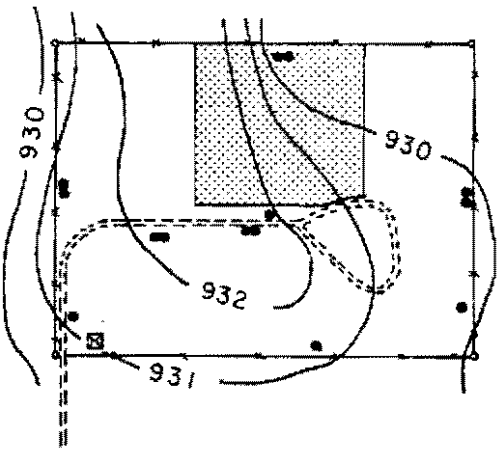
Arrows indicate the direction of groundwater flow.

Figure 23

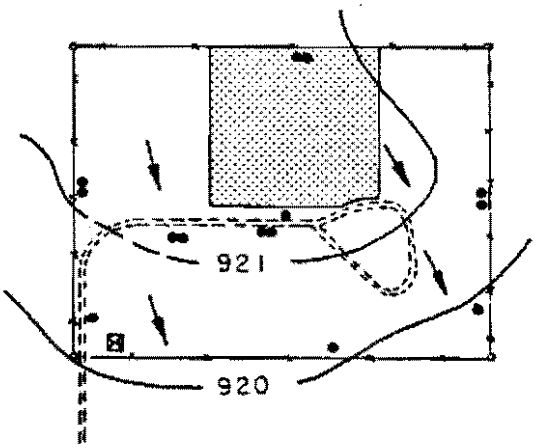
A PIEZOMETERS



B SURFACE TOPOGRAPHY



C WATER TABLE NOV. 1987



C.I. = 1.0 ft
0.304 m

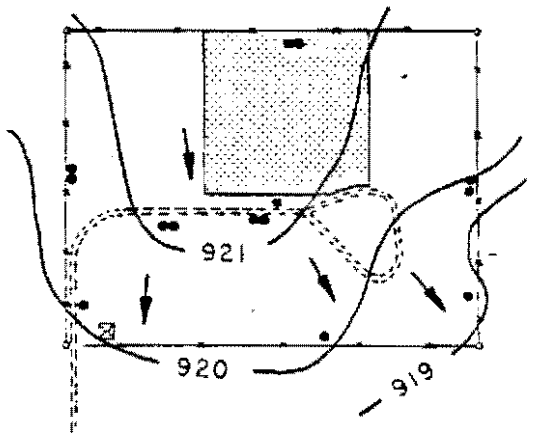
N

0 100 m

0 500 ft

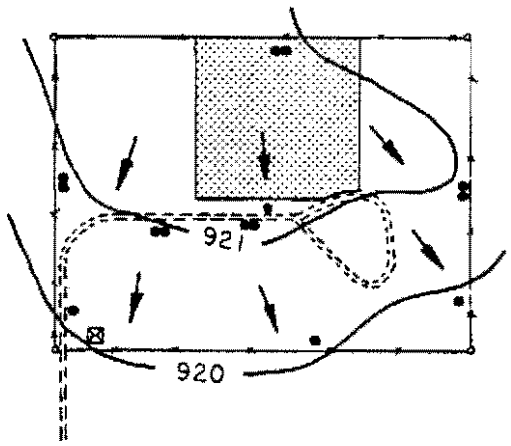
D

WATER TABLE
DEC. 1987



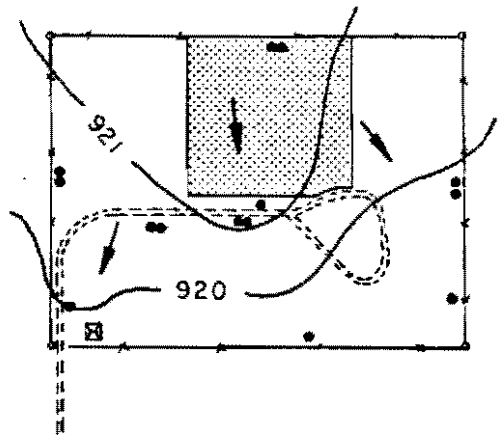
E

WATER TABLE
JAN. 1988



F

WATER TABLE
FEB. 1988



C.I. = 1.0 ft.
0.304 m

N

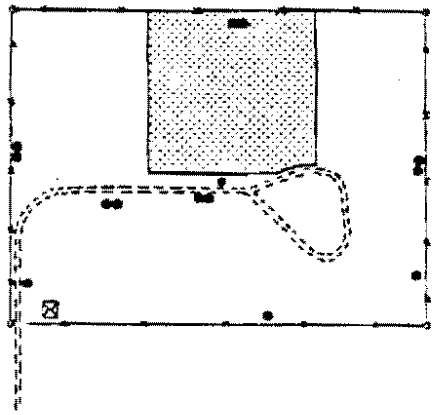
0 100 m

0 500 ft

G

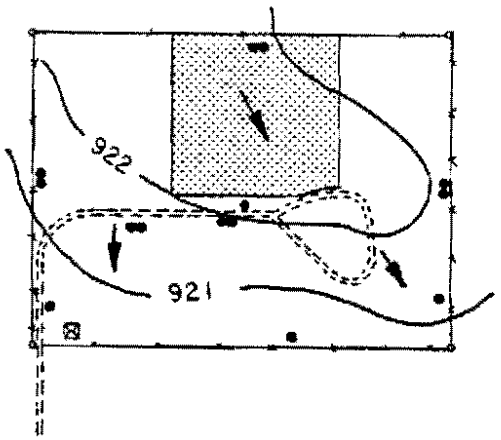
WATER TABLE
MAR. 1988

- NO WATER LEVELS TAKEN -



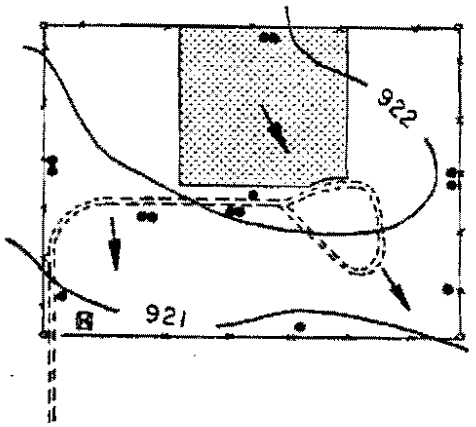
H

WATER TABLE
APR. 1988



I

WATER TABLE
MAY 1988



C.I. = 1.0 ft.
0.304 m

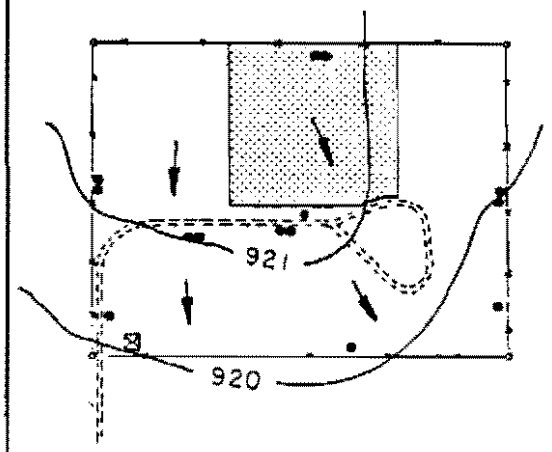
N

0 100 m

0 500 ft

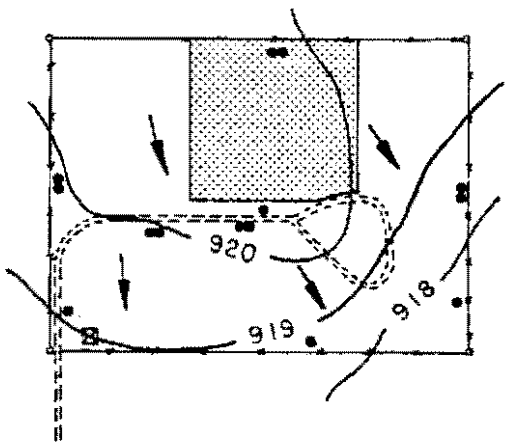
J

WATER TABLE
JUN. 1988



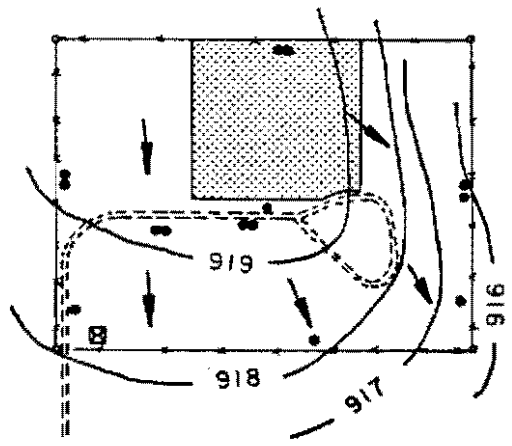
K

WATER TABLE
JUL. 1988



L

WATER TABLE
AUG. 1988



C.I. = 1.0 ft.
0.304 m

N

0 100 m

0 500 ft

APPENDIX IV

Textural Analyses

Sample	Depth(ft)	Description	% Sand	% Silt	% Clay
H1	2-4	Silty sand	50	39	11
H1	6.5	Sandy silt	30	61	9
H1	7-12	Sandy silt	40	57	3
H1	17-22	Sandy silt	25	73	2
H1	22-27	Sandy silt	24	73	3
H1	27-32	Sandy silt	17	79	4
H1	30-32	Clayey silt	3	62	35
H3	7-12	Silt	3	89	8
H3	12-17	Silt	9	87	4
H3	17-22	Sandy silt	19	78	3
H3	22-27	Sandy silt	19	79	2
H3	27-32	Sandy silt	18	80	2
H5	2-3	Silty sand	72	21	7
H5	3-6	Silty sand	53	41	6
H5	6-7	Clayey silt	17	55	28
H5	7.5-12	Sandy silt	35	58	7
H5	12-17	Silty sand	79	14	7
H5	17-22	Sand	92	8	0
H5	22-27	Sand	92	8	0
H7	2-6	Sandy silt	32	65	3
H7	7-8	Clayey silt	2	88	10
H7	12-17	Sandy silt	16	77	7
H7	17-22	Sandy silt	21	75	4
H7	22-27	Sandy silt	39	59	2
H7	27-32	Sandy silt	37	61	2

Sample	Depth(ft)	Description	% Sand	% Silt	% Clay
H9	0-1	Fill	29	62	9
H9	2-5	Sandy silt	34	55	11
H9	5-7.5	Sandy silt	27	69	4
H9	7.5-9	Sandy silt	14	82	4
H9	9-10	Sandy silt	16	80	4
H9	10-12.5	Silt	1	94	5
H9	12.5-13.5	Silt	2	90	8
H9	13.5-15	Sandy silt	12	79	9
H9	15-16	Sandy silt	10	83	7
H9	16-17.5	Sandy silt	29	67	4
H9	22.5-23.5	Sandy silt	25	71	4
H9	23.5-25	Sandy silt	18	78	4
H9	25-27.5	Silt	4	91	5
H9	30-40	Silty sand	77	23	0
H9	40-50	Silty sand	88	12	0
H9	55-60	Sand	90	8	2
H9	65-70	Sand	90	7	3
H9	70-75	Sand	91	7	2
H9	75-80	Sand	91	7	2
H10	0-2	Fill	38	46	14
H10	2-5	Silt	5	88	7
H10	5-7.5	Sandy silt	28	69	3
H10	7.5-8	Silt	6	93	1
H10	8-10	Silt	8	88	4
H10	10-11	Sandy silt	48	51	1
H10	11-12	Sandy silt	47	51	2
H10	12-13	Sandy silt	44	47	9
H10	13-15	Silty sand	77	23	0
H10	15-16	Sand	92	8	0
H10	16-17	Clayey silt	4	86	10
H10	20-25	Silty sand	83	14	3
H11	@17	Silty sand	53	42	5

APPENDIX V

Water Analyses

Total A	Total alkalinity ($CaCO_3$) in milligrams/litre
As	Arsenic in micrograms/litre
HCO_3	Bicarbonate in milligrams/litre
Cd	Cadmium in micrograms/litre
CO_3	Carbonate in milligrams/litre
Cl	Chloride in milligrams/litre
F	Fluoride in milligrams/litre
Total H	Total hardness in milligrams/litre
Pb	Lead in micrograms/litre
NO_3	Nitrate reported as N in milligrams/litre
pH	Field pH
Temp.	Field temperature in degrees Celsius
Se	Selenium in micrograms/litre
% Na	Percent sodium
SO_4	Sulfate in milligrams/litre
TDS	Total Dissolved Solids in milligrams/litre
Turb.	Turbidity
SAR	Sodium absorption ratio
Cond.	Specific conductance in micromhos/cm
Ba	Barium in micrograms/litre
Ca	Calcium in milligrams/litre
Cu	Copper in micrograms/litre
Fe	Iron in milligrams/litre
Mg	Magnesium in milligrams/litre
Mn	Manganese in milligrams/litre
K	Potassium in milligrams/litre
Na	Sodium in milligrams/litre
Cr	Chromium in micrograms/litre
Zn	Zinc in micrograms/litre

	Piezometer				
	H1	H2	H3	H4	H5
Total A	185	646	291	390	272
As	3.1	1.1	1.9	1.2	5.8
HCO3	226	789	355	476	332
Cd	2.26	2.11	0.78	0.66	1.51
CO3	____	____	____	____	____
Cl	2.1	10.7	2.3	2.2	3.6
F	0.2	0.1	0.2	0.8	0.3
Total H	212	704	292	430	273
Pb	1.3	1.6	5.1	0.7	0.1
NO3	____	5.0	____	1.2	0.1
pH	7.75	7.15	7.79	7.75	7.75
Temp.	7	7	6	7	7
Se	1.0	1.0	____	____	____
% Na	2.1	1.8	3.2	3.8	5.6
SO4	23	24	9.0	42	9.0
TDS	214	682	289	423	276
Turb.	2.0	3.0	< 1	2.0	2.0
SAR	0.06	0.10	0.11	0.17	0.20
Cond.	351	1023	463	647	459
Ba	317	138	227	112	177
Ca	55	146	64.8	63.2	57.1
Cu	4.7	4.7	6.1	0.7	2.4
Fe	0.136	0.215	0.033	0.387	0.009
Mg	18.1	82.5	31.5	66.1	31.7
Mn	0.422	0.359	0.475	0.021	0.728
K	1.90	2.10	2.50	1.60	2.80
Na	2.10	6.10	4.50	7.90	7.50
Cr	____	12.7	1.90	____	4.50
Zn	53	138	70	154	80

	Piezometer				
	H6	H7	H8	H9	H10
Total A	530	292	X	287	245
As	1.1	1.4	X	3.9	1.3
HCO3	647	356	X	350	299
Cd	1.0	1.0	X	1.24	0.41
CO3			X		
Cl	6.8	1.7	X	18.0	5.3
F	0.1	0.3	X	0.2	0.2
Total H	554	291	X	325	307
Pb	0.9	0.5	X	0.7	0.6
NO3	0.2	0.2	X		10.2
pH	7.61	7.78	X	7.75	7.71
Temp.		8	X	8	8
Se	1.0		X		4.0
% Na	1.7	2.8	X	4.5	2.9
SO4	16	10	X	33	30
TDS	537	287	X	343	334
Turb.	2.0	< 1	X	5.0	2.0
SAR	0.08	0.10	X	0.17	0.11
Cond.	829	461	X	527	487
Ba	158	229	X	239	256
Ca	136	56.7	X	76.5	64.0
Cu	5.2	5.2	X	1.4	2.2
Fe	0.240	0.047	X	0.685	0.254
Mg	52	36.2	X	32.6	35.7
Mn	0.124	0.828	X	0.527	0.031
K	2.7	2.1	X	3.0	1.8
Na	4.5	3.9	X	7.1	4.3
Cr			X	9.8	6.7
Zn	135	16	X	76	28

	Piezometer				
	H11	H12	H13	H14	H15
Total A	475	307	277	452	264
As	1.2	1.4	1.6	1.5	1.6
HCO3	580	375	338	552	322
Cd	0.38	0.24	1.26	0.41	0.71
CO3	____	____	____	____	____
Cl	247	18.0	6.4	2.3	1.4
F	0.2	0.3	0.3	0.4	0.2
Total H	699	385	331	464	299
Pb	0.5	0.2	0.6	0.3	1.2
NO3	1.4	9.3	5.9	____	1.1
pH	7.46	7.0	7.81	7.75	7.73
Temp.	8	7	8	7	6
Se	1.0	____	____	____	____
% Na	21.3	1.6	2.9	0.6	2.1
SO4	65	40	20	8.0	10.0
TDS	909	413	325	419	277
Turb.	2.0	2.0	< 1	< 1	2.0
SAR	1.44	0.06	0.11	0.03	0.08
Cond.	1394	613	515	680	450
Ba	447	146	104	135	88.8
Ca	107	75.2	48.4	54.1	64.4
Cu	3.0	____	0.50	4.9	1.10
Fe	0.250	0.189	0.069	0.186	1.24
Mg	105	47.8	51.0	79.8	33.5
Mn	0.014	0.037	0.013	0.021	0.033
K	5.50	2.50	1.90	1.80	1.70
Na	87.4	2.90	4.60	1.30	3.00
Cr	6.70	____	2.70	1.00	____
Zn	70	80	55	111	130

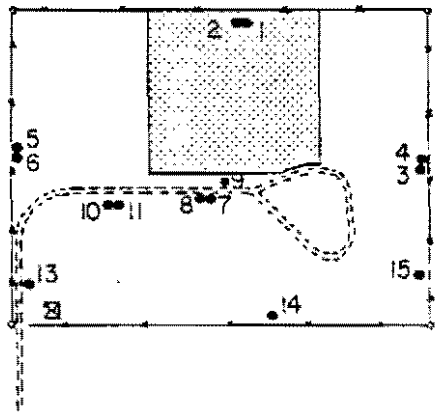
APPENDIX VI

Isoconcentration Maps of Selected Parameters from within the Saturated Zone

Background levels were obtained from piezometer H12 located approximately 1000 feet (304.8 m) to the northwest of the landfill (Figure 7). Recommended concentration limits (RCL) are established by the Environmental Protection Agency based upon taste and esthetic appearance. Maximum permissable concentrations (MPC) are similarly based upon health effects (Table 1).

Figure 24

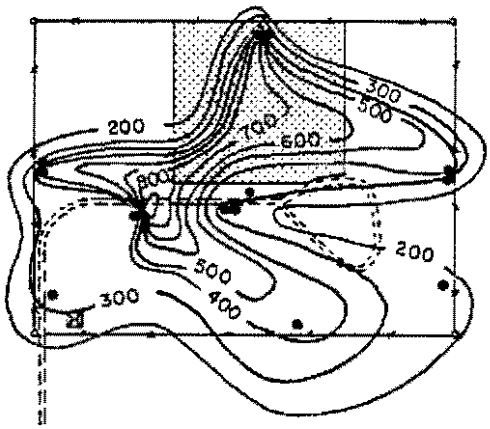
A WATER SAMPLING STATION
(PIEZOMETERS)



B TOTAL DISSOLVED SOLIDS

Background Level = 413 mg/L
RCL = 500 mg/L

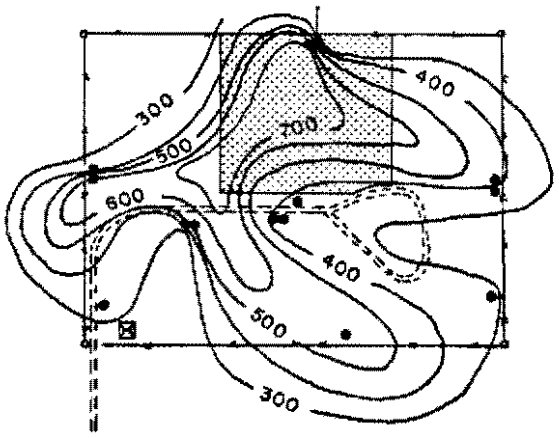
C.I. = 100 mg/L

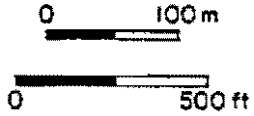


C BICARBONATE

Background Level = 375 mg/L

C.I. = 100 mg/L





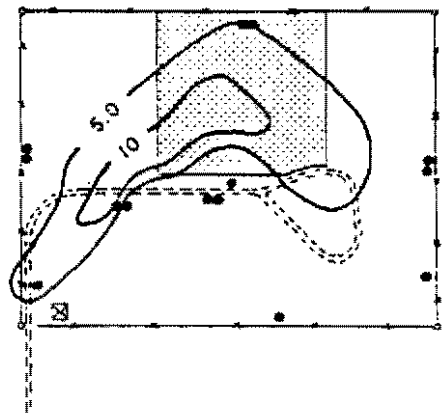
D

NITRATE (N)

Background Level = 9.3 mg/L

RCL = 10 mg/L

C.I. = 5.0 mg/L

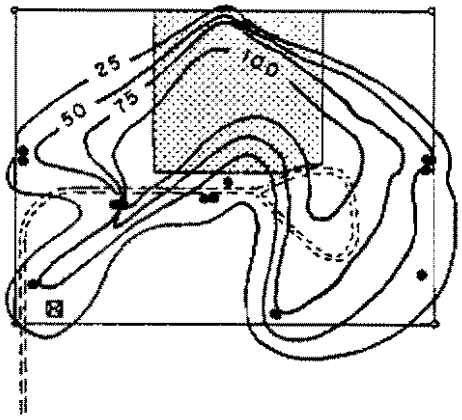


E

MAGNESIUM

Background Level = 47.8 mg/L

C.I. = 25 mg/L



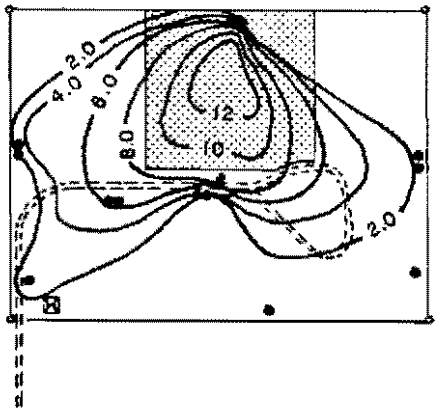
F

CHROMIUM

Background Level = <1.0ug/L

MPC = 50 ug/L

C.I. = 2.0 ug/L



N

0 100 m

0 500 ft

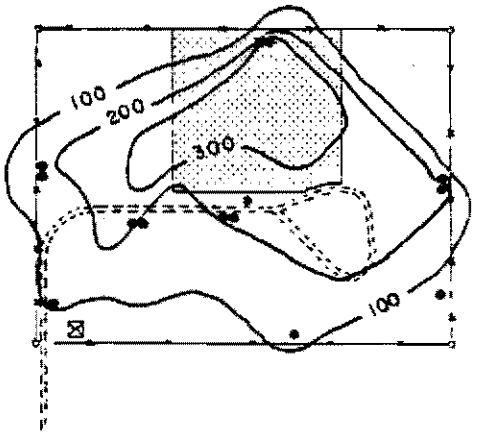
G

BARIUM

Background Level = 146 ug/L

MPC = 1000 ug/L

C.I. = 100 ug/L



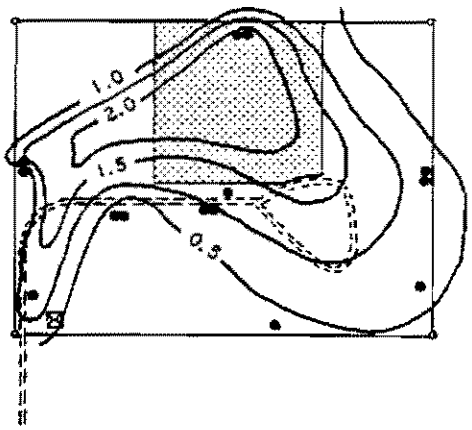
H

CADMIUM

Background Level = 0.24 ug/L

MPC = 10 ug/L

C.I. = 0.50 ug/L



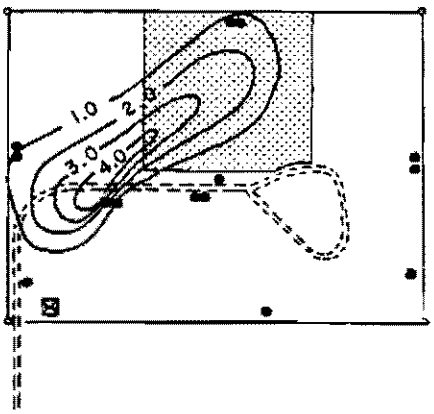
I

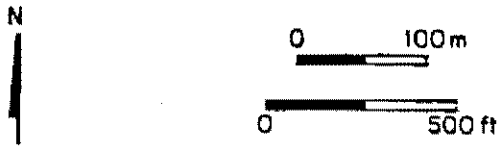
SELENIUM

Background Level = 1.0 ug/L

MPC = 10 ug/L

C.I. = 1.0 ug/L



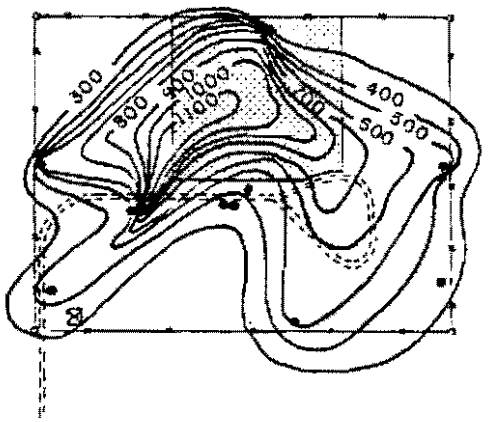


J

FIELD CONDUCTIVITY

Background Level =486 umhos/cm

C.I. = 100 umhos/cm

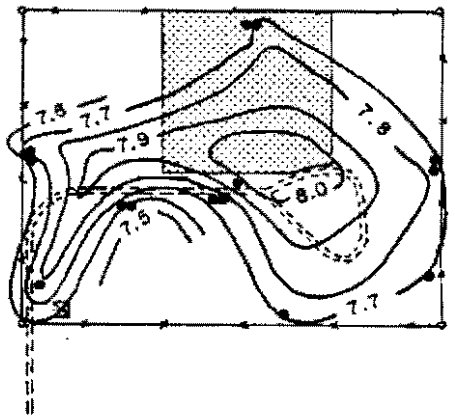


K

FIELD pH

Background Level = 7.75

C.I. = 0.10

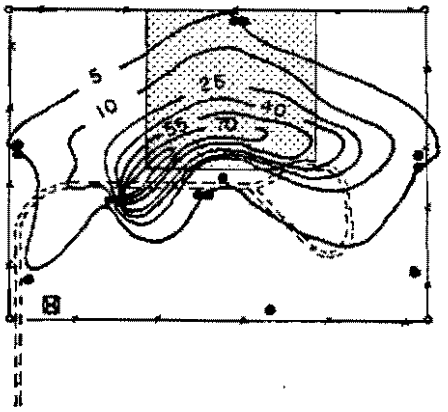


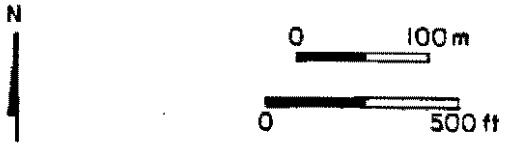
L

SODIUM

Background Level = 2.9 mg/L

C.I. = 5 mg/L & 15 mg/L



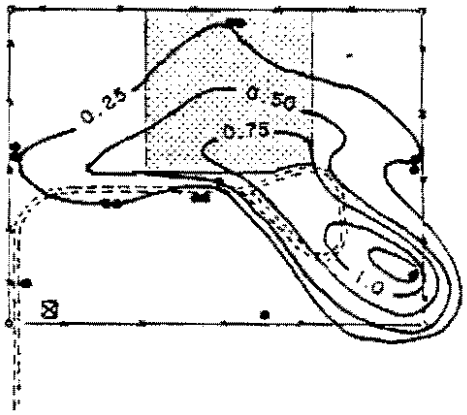


M

IRON

Background Level = 0.19 mg/L
RCL = 0.30 mg/L

C.I. = 0.25 mg/L

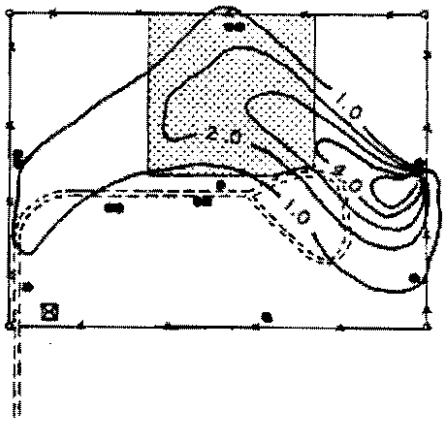


N

LEAD

Background Level = 0.2 ug/L
MPC = 50 ug/L

C.I. = 1.0 ug/L

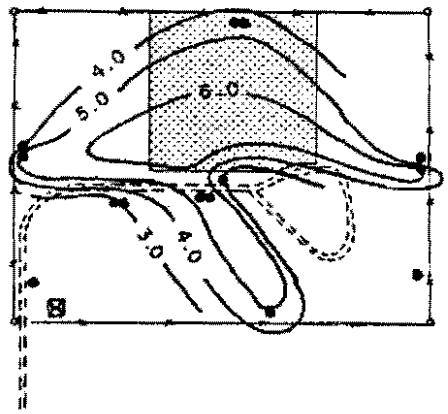


O

COPPER

Background Level = <1.0 ug/L
RCL = 1000 ug/L

C.I. = 1.0 ug/L



N

0 100 m

0 500 ft

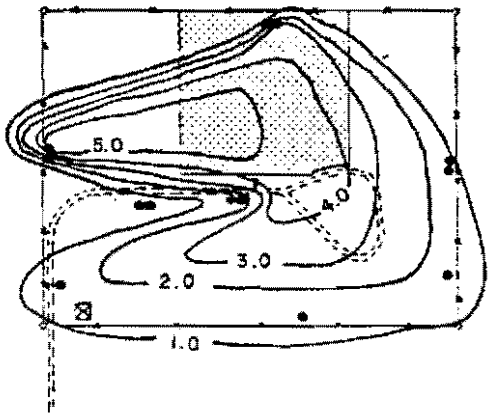
P

ARSENIC

Background Level = 1.4 ug/L

MPC = 50 ug/L

C.I. = 1.0 ug/L



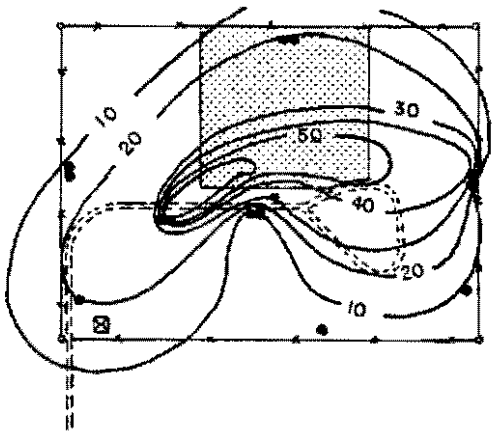
Q

SULFATE

Background Level = 40 mg/L

RCL = 250 mg/L

C.I. = 10 mg/L



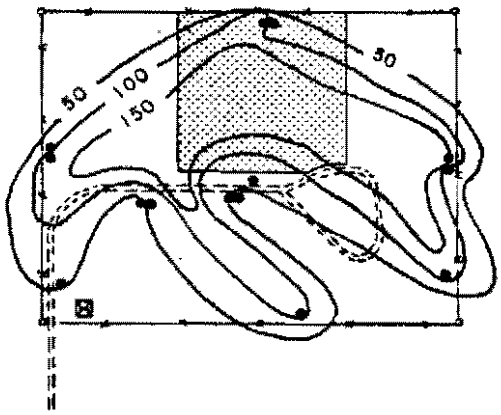
R

ZINC

Background Level = 80 ug/L

RCL = 5000 ug/L

C.I. = 50 ug/L

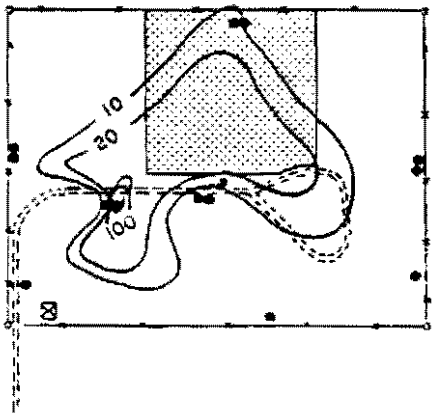


S

CHLORIDE

Background Level = 18 mg/L
RCL = 250 mg/L

C.I. = 10 mg/L & 100 mg/L

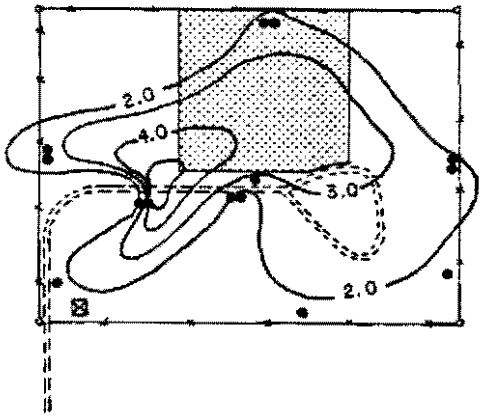


T

POTASSIUM

Background Level = 2.5 mg/L

C.I. = 1.0 mg/L

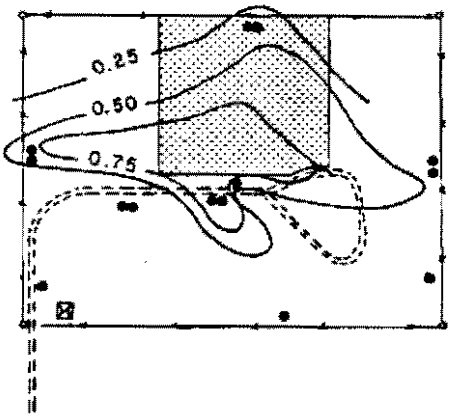


U

MANGANESE

Background Level = 0.037 mg/L
RCL = 0.05 mg/L

C.I. = 0.25 mg/L



N

0 100 m

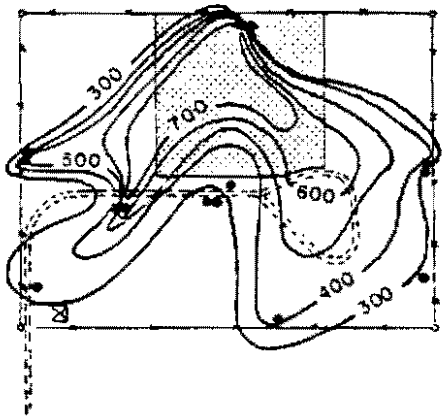
0 500 ft

V

TOTAL HARDNESS

Background Level = 385 mg/L

C.I. = 100 mg/L

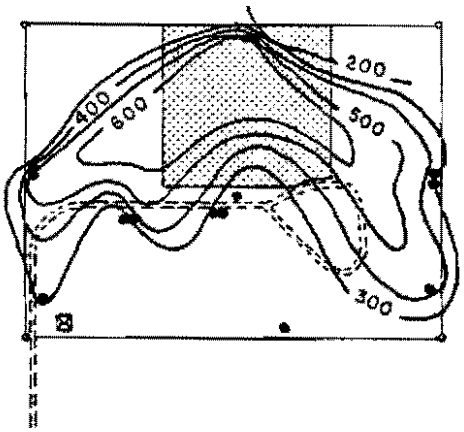


W

TOTAL ALKALINITY

Background Level = 307 mg/L

C.I. = 100 mg/L

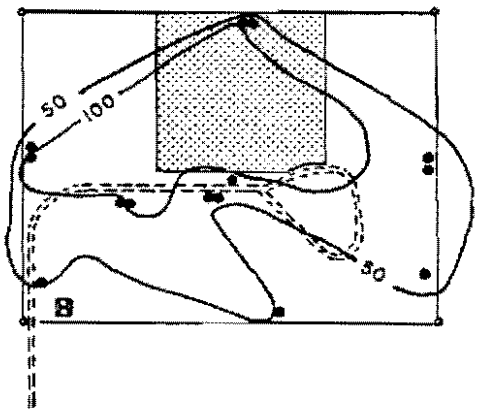


X

CALCIUM

Background Level = 75 mg/L

C.I. = 50 mg/L



N

0 100 m

0 500 ft

APPENDIX VII

Hydraulic Conductivity Estimates Using Grain-size Analyses

K = hydraulic conductivity, a measure of the ability of a material to transmit fluid.

σi = inclusive standard deviation, a statistical measure of the dispersion about the median diameter for a given grain-size distribution.

d16 = grain-size diameter at which 16 percent by weight of the sample is finer and 84 percent is coarser (similar for d5, d84, and d95).

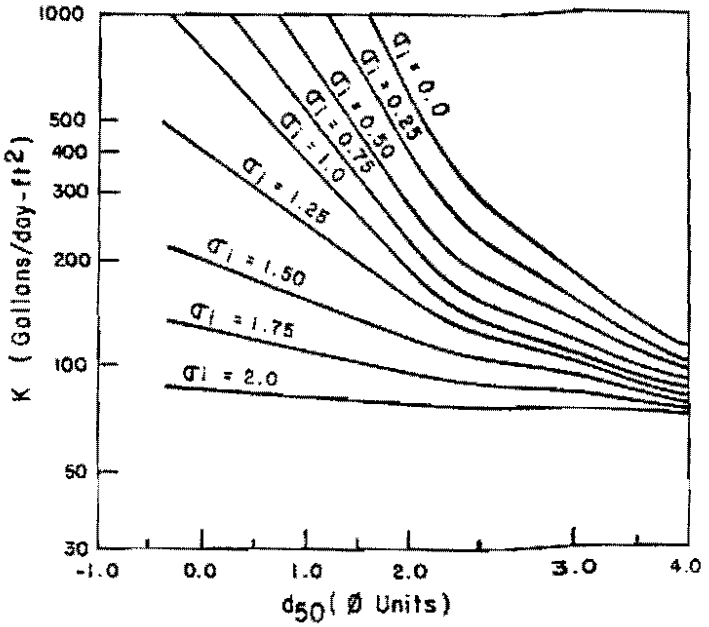
d50 = the median grain-size.

Figure 25

From MASCH & DENNY, (1966):

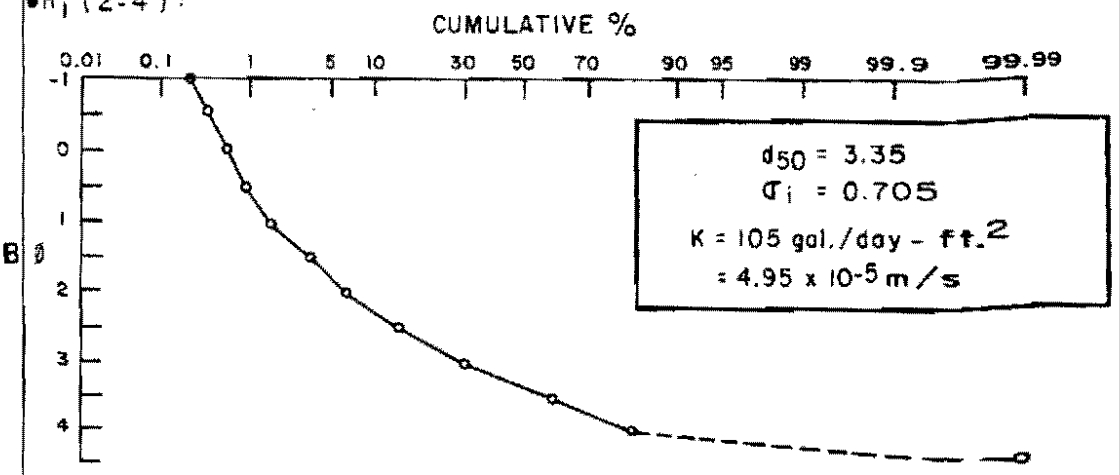
A

$$\sigma_i = \frac{d_{16} - d_{84}}{4} + \frac{d_5 - d_{95}}{6.6}$$



B

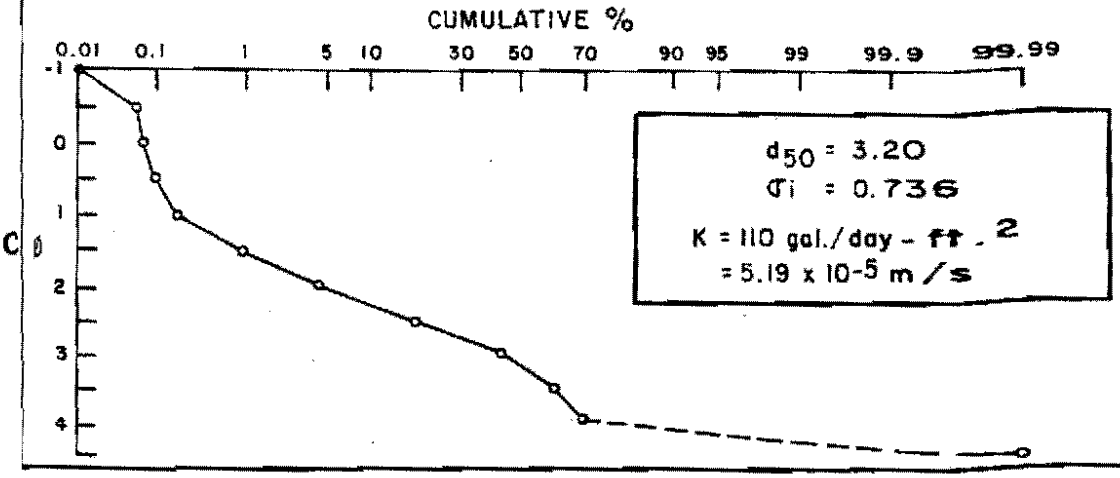
●H1 (2-4'):



d_{50} = 3.35
σ_i = 0.705
K = 105 gal./day - ft.²
= 4.95 x 10⁻⁵ m/s

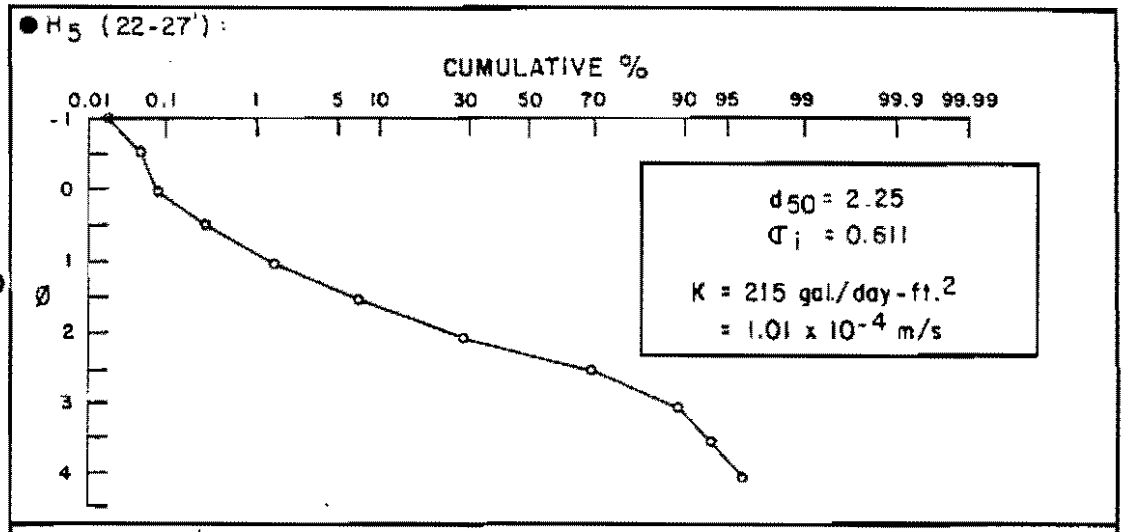
C

●H5 (3-6'):

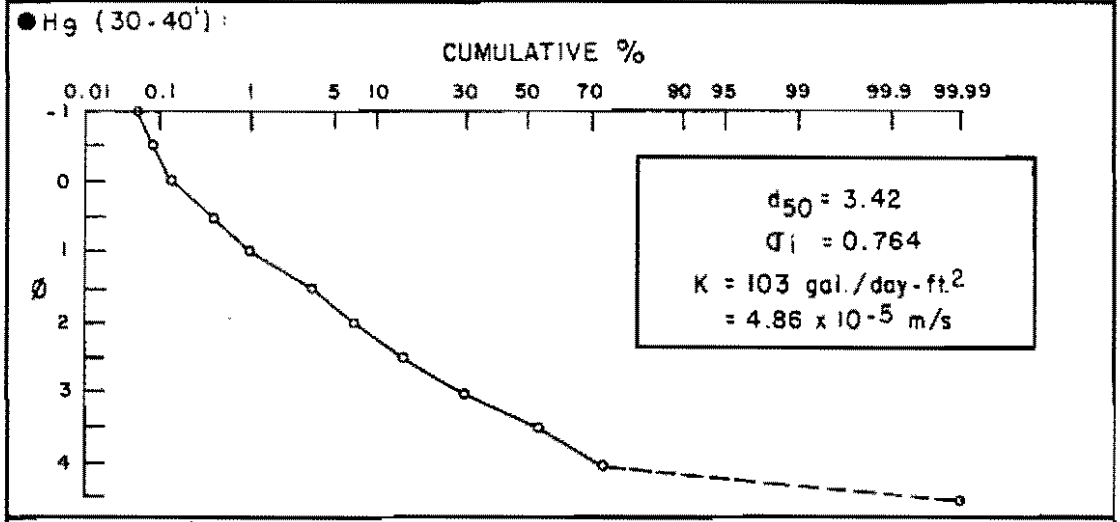


d_{50} = 3.20
σ_i = 0.736
K = 110 gal./day - ft.²
= 5.19 x 10⁻⁵ m/s

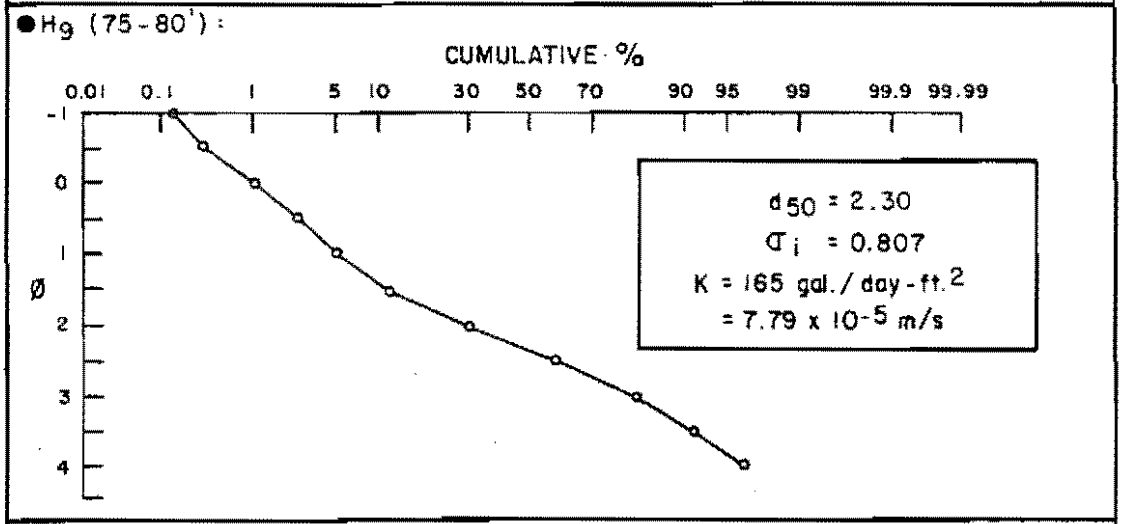
D



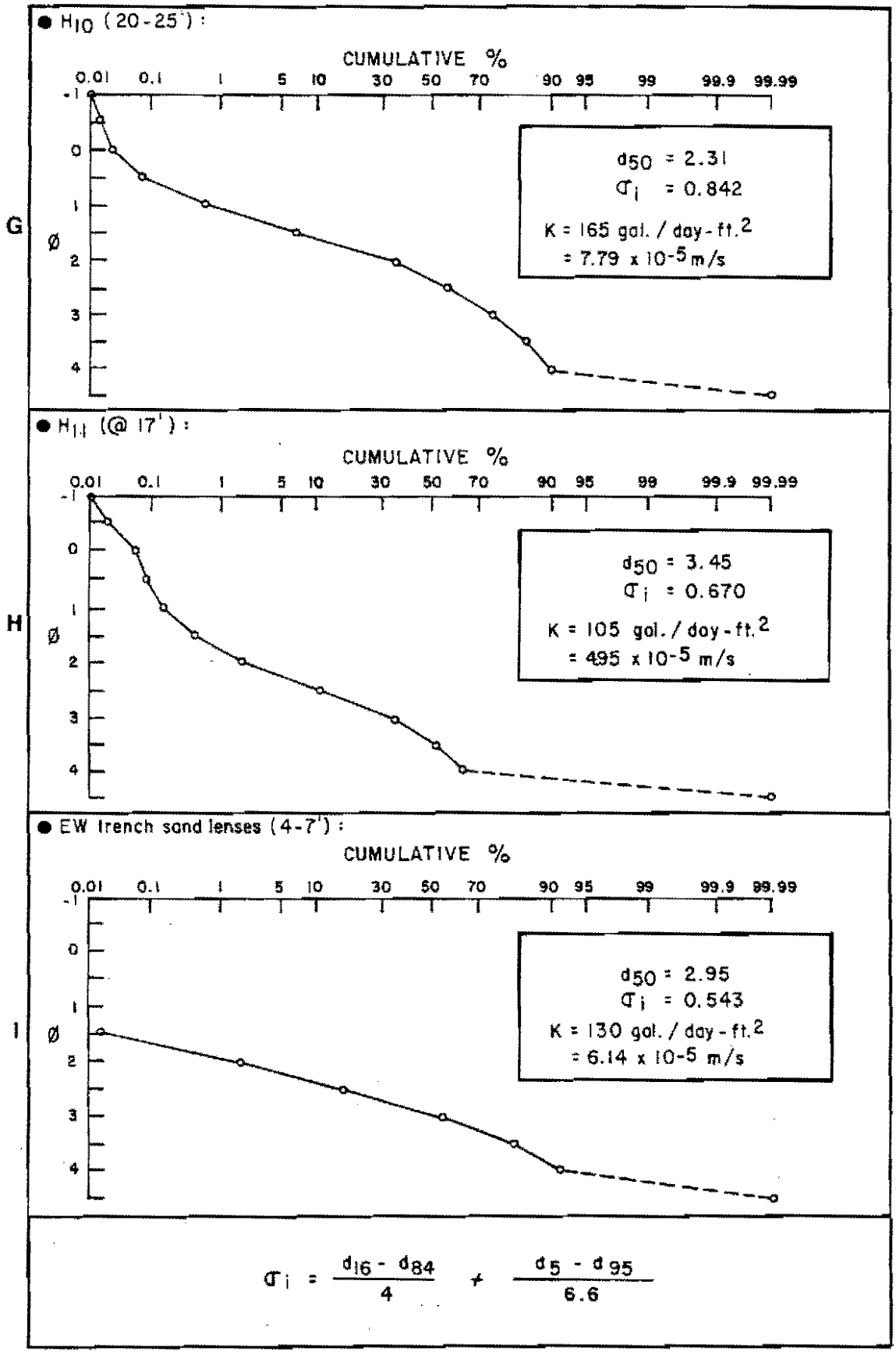
E

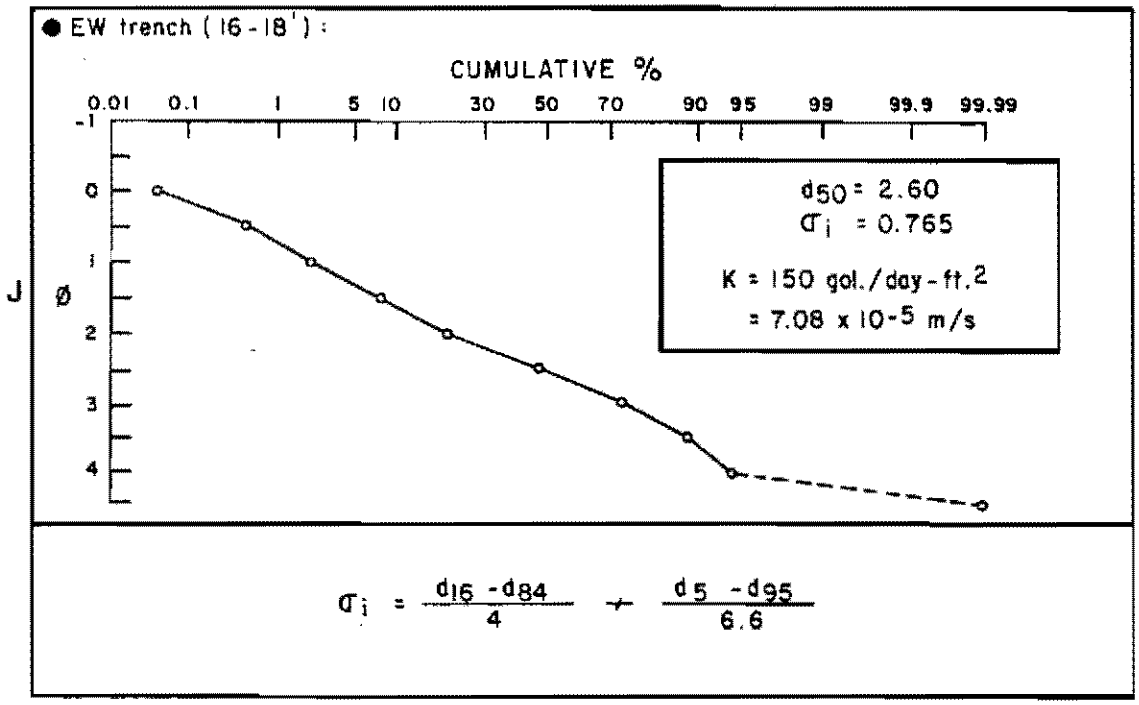


F



$$\sigma_i = \frac{d_{16} - d_{84}}{4} + \frac{d_5 - d_{95}}{6.6}$$


● H10 (20-25'):
CUMULATIVE %
0.01 0.1 1 5 10 30 50 70 90 95 99 99.9 99.99
-1 0 1 2 3 4
Ø
G
d_{50} = 2.31
σ_i = 0.842
K = 165 gal. / day - ft.^{2}
= 7.79 x 10^{-5} m/s
● H11 (@ 17'):
CUMULATIVE %
0.01 0.1 1 5 10 30 50 70 90 95 99 99.9 99.99
-1 0 1 2 3 4
Ø
H
d_{50} = 3.45
σ_i = 0.670
K = 105 gal. / day - ft.^{2}
= 495 x 10^{-5} m/s
● EW trench sand lenses (4-7'):
CUMULATIVE %
0.01 0.1 1 5 10 30 50 70 90 95 99 99.9 99.99
-1 0 1 2 3 4
Ø
I
d_{50} = 2.95
σ_i = 0.543
K = 130 gal. / day - ft.^{2}
= 6.14 x 10-5 m/s
$\sigma_i = \frac{d_{16} - d_{84}}{4} + \frac{d_5 - d_{95}}{6.6}$


● EW trench (16 - 18') :
CUMULATIVE %
0.01 0.1 1 5 10 30 50 70 90 95 99 99.9 99.99
-1
0
1
2
3
4
J
ø
d_{50} = 2.60
σ_i = 0.765
K = 150 gal./day-ft.^{2}
= 7.08 x 10^{-5} m/s
$\sigma_i = \frac{d_{16} - d_{84}}{4} + \frac{d_5 - d_{95}}{6.6}$

APPENDIX VIII

Apparent and Interpreted Resistivity Profiles

Each resistivity station profile includes the field curve plotted as apparent resistivity versus electrode spacing, the depths and resistivities obtained by automatic interpretation, lithology, and water table level. A lithologic column is presented for each resistivity station and corresponds to the station that it is adjacent to. Piezometer location in relation to resistivity stations are shown in the inset taken from Figure 12.

LEGEND

TOPSOIL / FILL

SAND

SANDY SILT

CLAY

SILTY CLAY

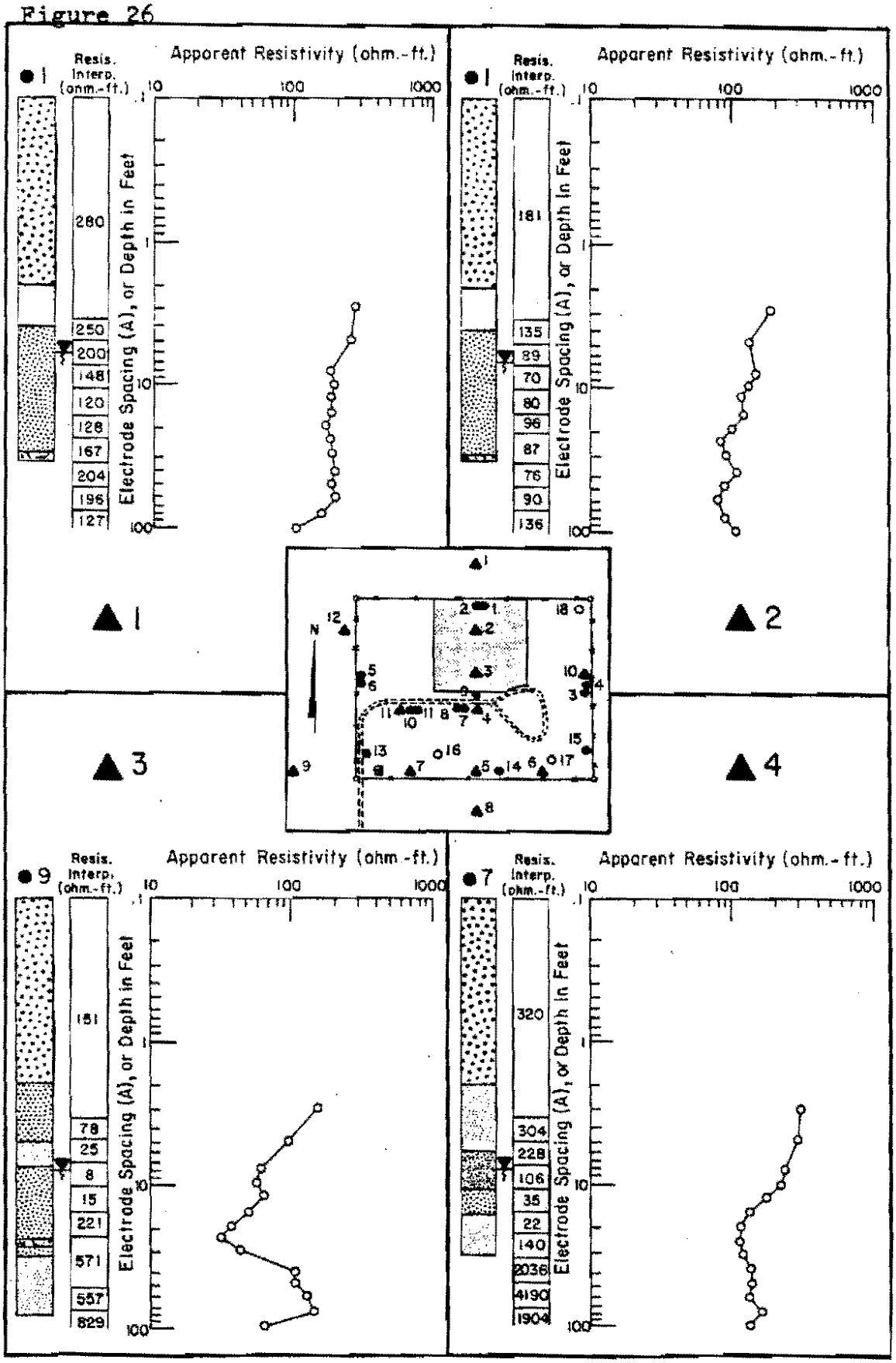
● 1 PIEZOMETER

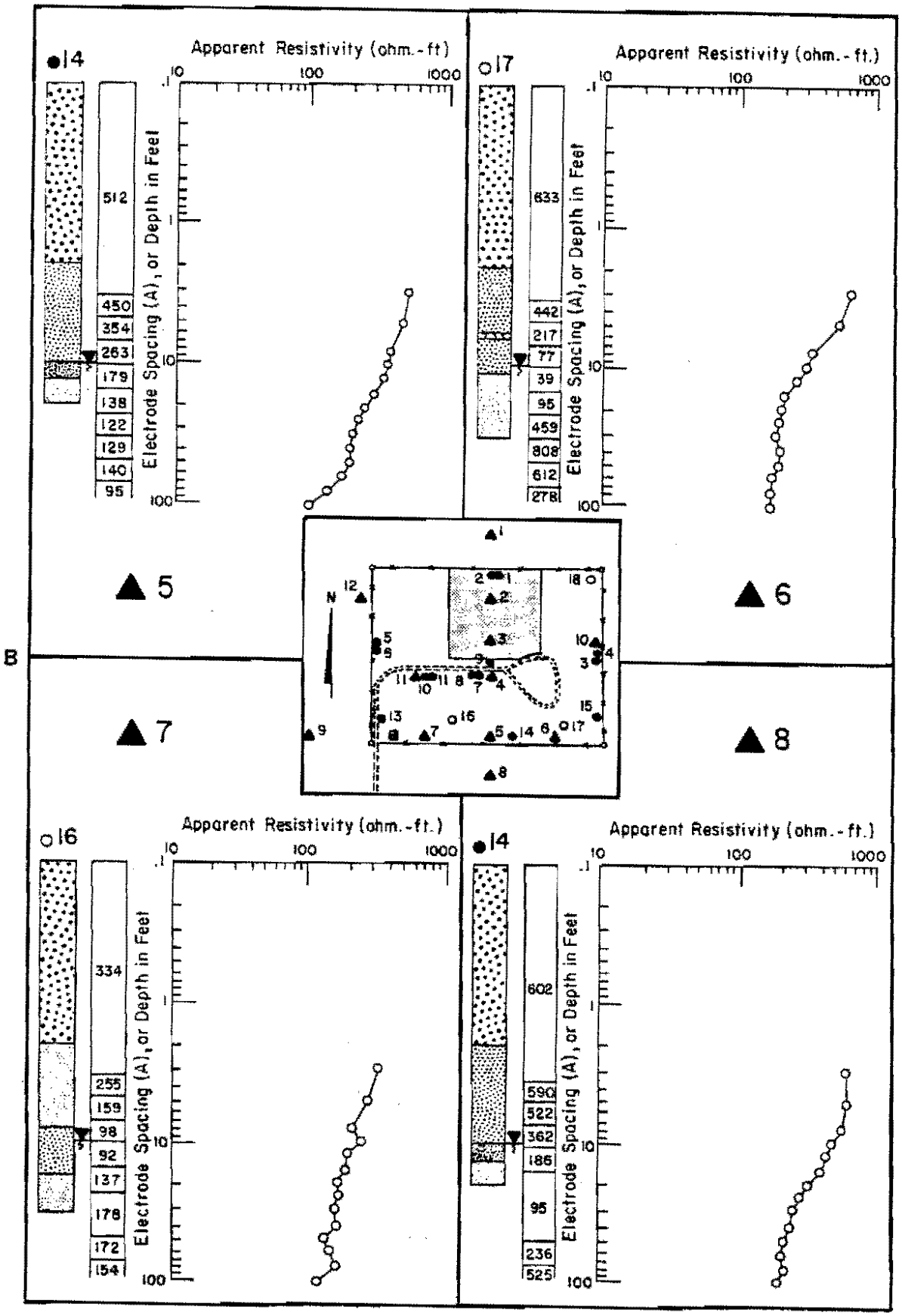
▲ 3 RESISTIVITY STATION

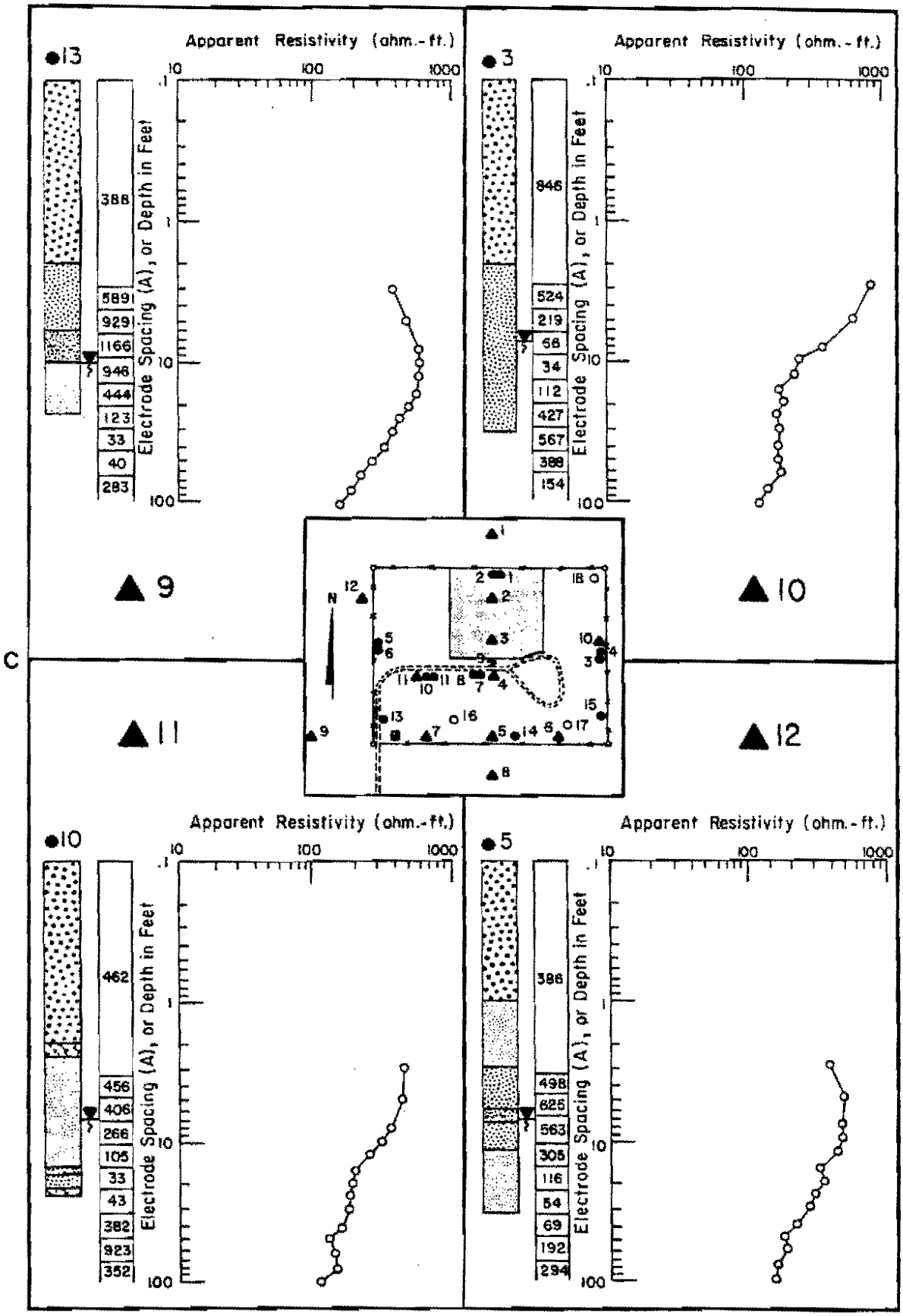
○ 16 TEST HOLE

WATER LEVEL

Figure 26


A
Resis. Interp. (ohm.-ft.)
Apparent Resistivity (ohm.-ft.)
Electrode Spacing (A), or Depth in Feet
10
100
1000
1
10
100
280
250
200
148
120
128
167
204
196
127
181
135
89
70
80
96
87
76
90
136
320
304
228
106
35
22
140
2036
4190
1904
151
78
25
8
15
221
571
557
829
1
2
3
4
N
12
18
9
16
17
14

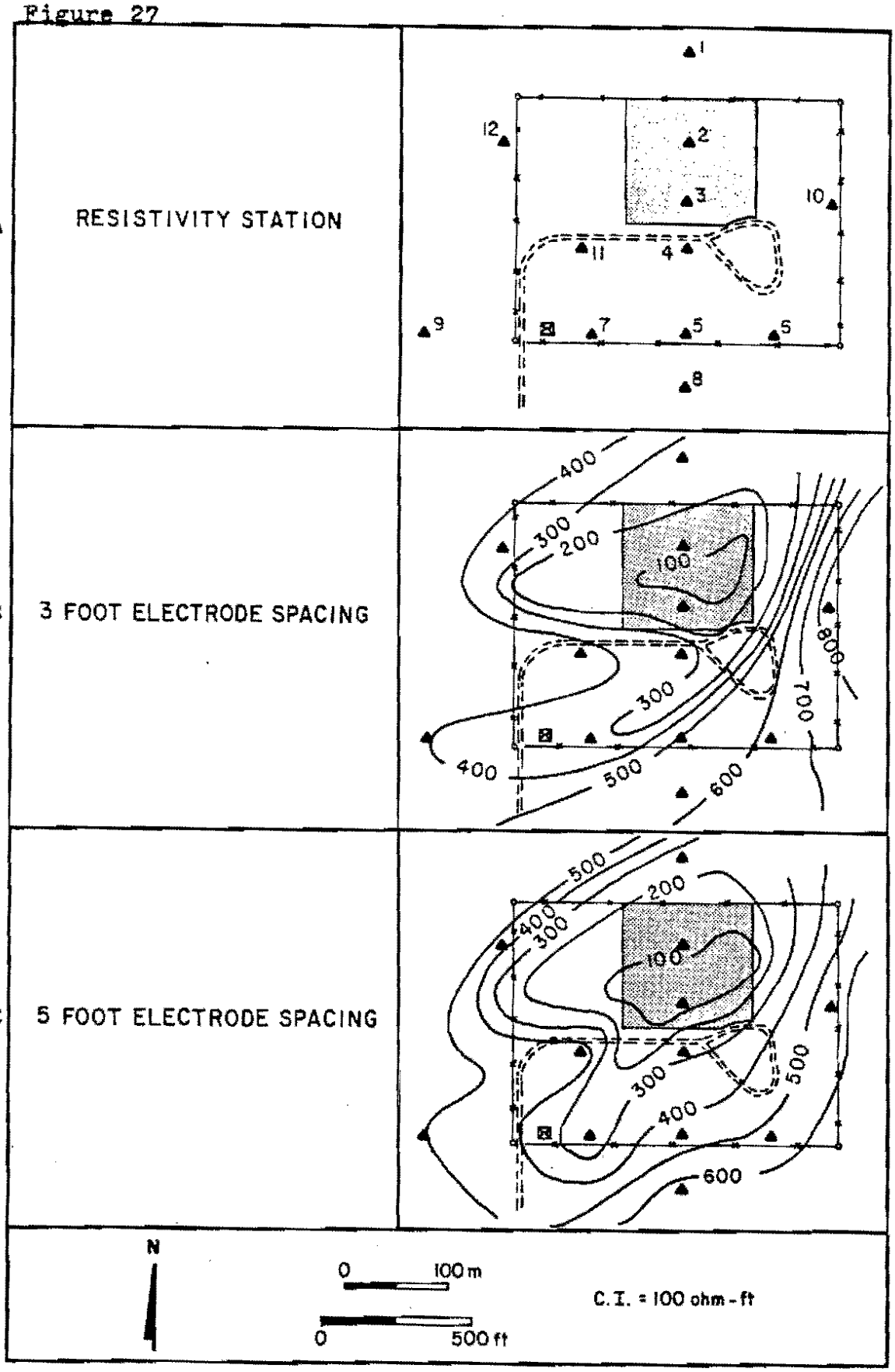

●14
Apparent Resistivity (ohm.-ft)
Electrode Spacing (A), or Depth in Feet
512
450
354
263
179
138
122
129
140
95
○17
633
442
217
77
39
95
459
808
612
278
▲5
▲6
▲7
▲8
B
N
○16
334
255
159
98
92
137
178
172
154
602
590
522
362
186
95
236
525


Apparent Resistivity (ohm.-ft.)
Electrode Spacing (A), or Depth in Feet
10
100
1000
.1
1
10
100
●13
388
589
929
1166
946
444
123
33
40
283
▲9
●3
846
524
219
68
34
112
427
567
388
154
▲10
C
N
▲11
▲12
●10
462
456
406
266
105
33
43
382
923
352
●5
386
498
625
563
305
116
54
69
192
294

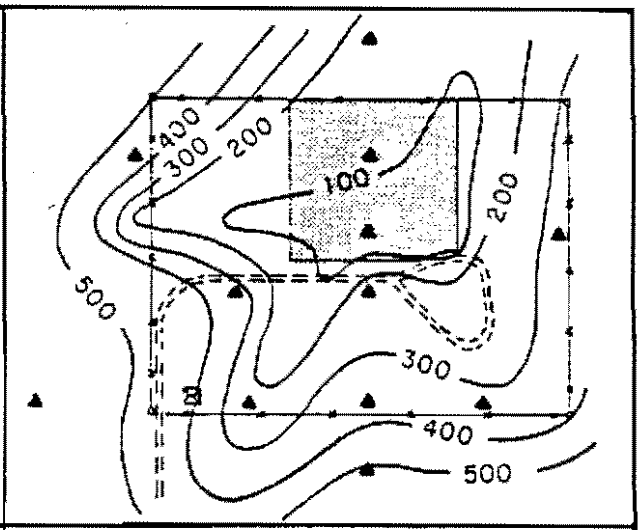
APPENDIX IX

Apparent Isoresistivity Maps

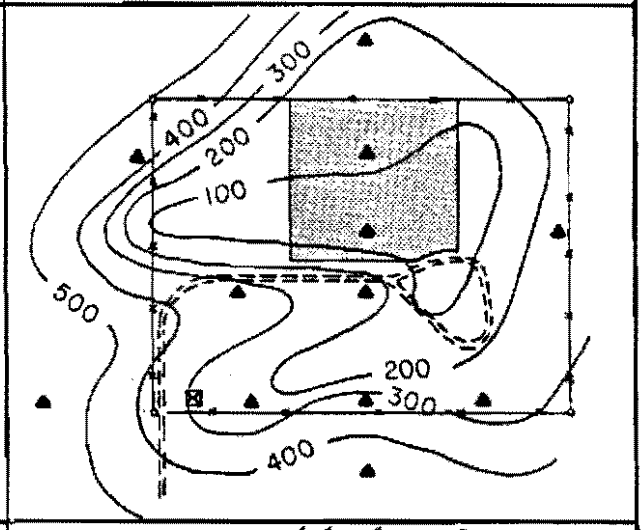
Figure 27


A
RESISTIVITY STATION
1
2
3
4
5
6
7
8
9
10
11
12
B
3 FOOT ELECTRODE SPACING
100
200
300
400
500
600
700
800
C
5 FOOT ELECTRODE SPACING
100
200
300
400
500
600
N
0
100 m
0
500 ft
C.I. = 100 ohm-ft

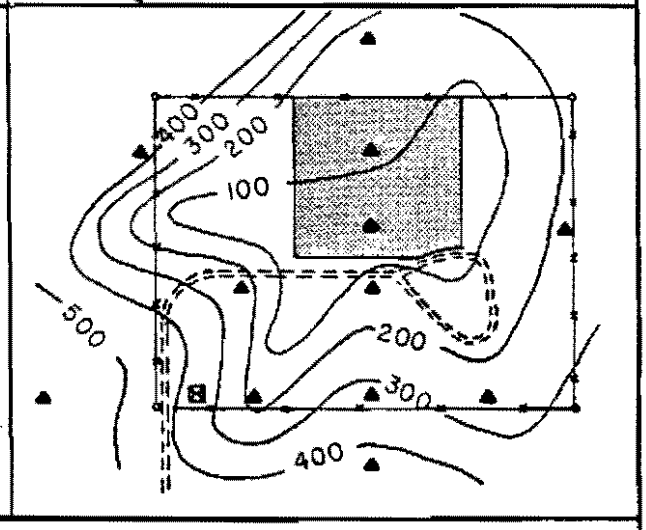
D 8 FOOT ELECTRODE SPACING



E 10 FOOT ELECTRODE SPACING



F 12 FOOT ELECTRODE SPACING

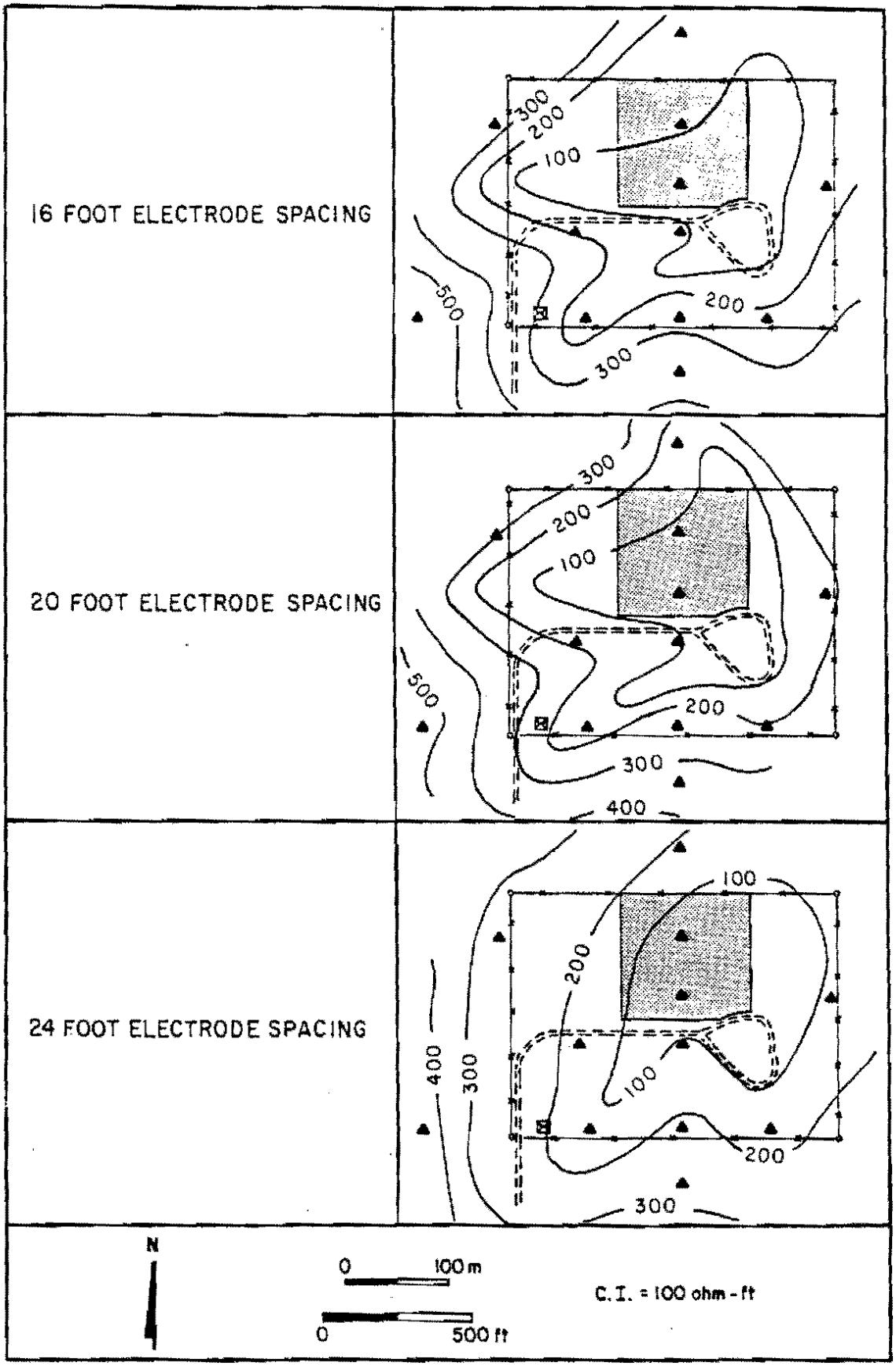


N

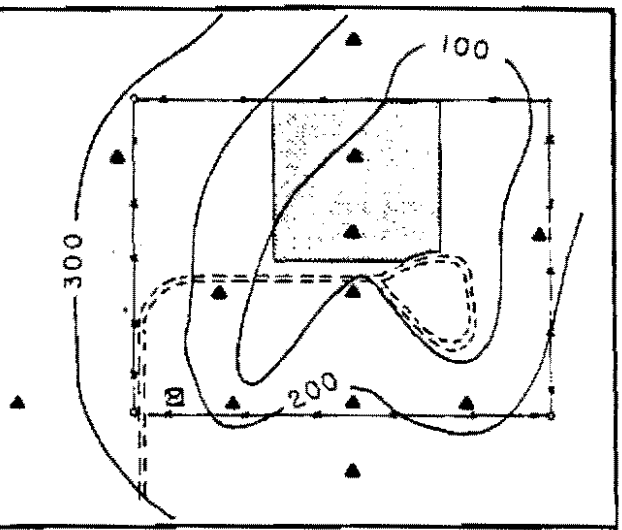
0 100 m

0 500 ft

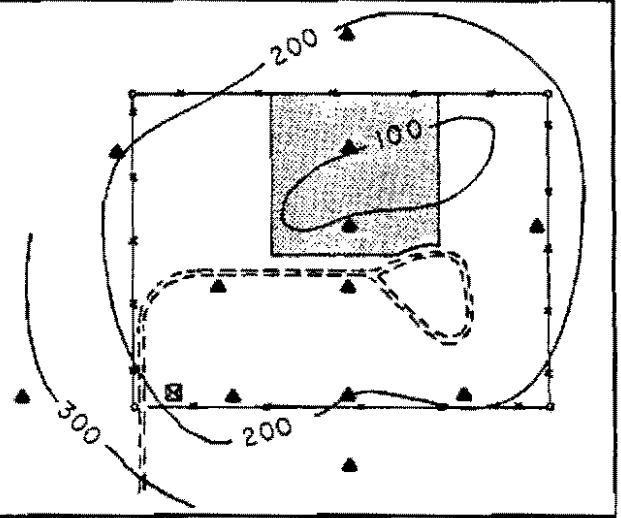
C.I. = 100 ohm - ft


G
16 FOOT ELECTRODE SPACING
300
200
100
500
200
300
H
20 FOOT ELECTRODE SPACING
300
200
100
500
200
300
400
I
24 FOOT ELECTRODE SPACING
100
200
400
300
100
200
300
N
0
100 m
0
500 ft
C.I. = 100 ohm - ft

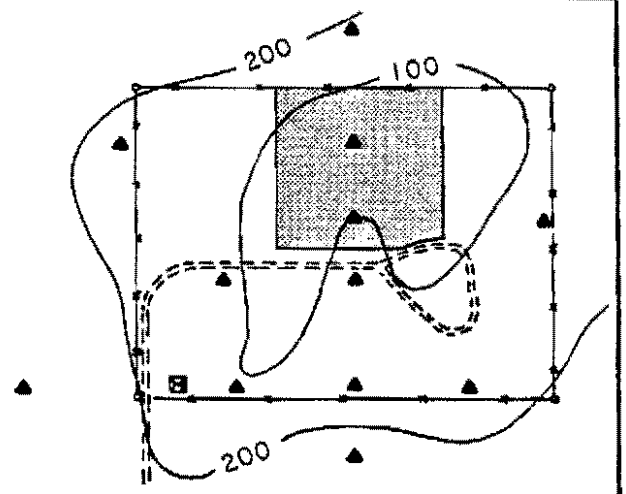
J 30 FOOT ELECTRODE SPACING



K 40 FOOT ELECTRODE SPACING



L 50 FOOT ELECTRODE SPACING



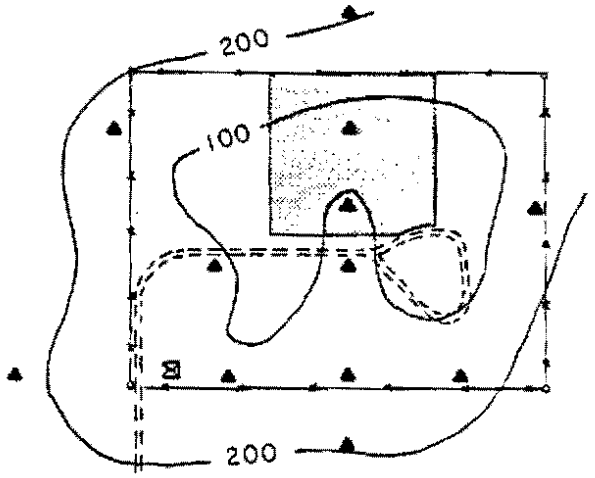
N

0 100 m

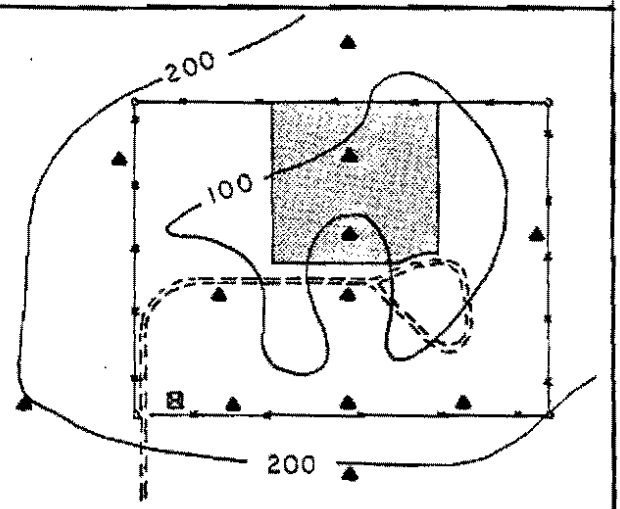
0 500 ft

C. I. = 100 ohm - ft

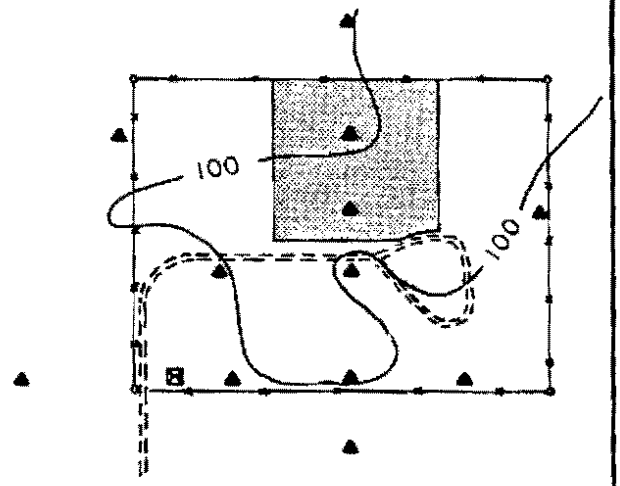
M 60 FOOT ELECTRODE SPACING



N 80 FOOT ELECTRODE SPACING



O 100 FOOT ELECTRODE SPACING



N

0 100 m

0 500 ft

C.I. = 100 ohm-ft

REFERENCES

Andersen, John, and Dornbush, James, 1967, Influence of sanitary landfill on groundwater quality: Journal of the American Water Works Association, v. 59, No. 4, p. 457-470.

Anderson, M.P., 1984, Movement of contaminants in groundwater: Groundwater transport-advection and dispersion: Studies in Geophysics-Groundwater Contamination, Washington, D.C., National Academy Press, p. 37-45.

Baedecker, Mary Jo, and Back, William, 1979, Hydrogeological processes and chemical reactions at a landfill: Groundwater, v. 17, No. 5, p. 429-437.

Baedecker, Mary Jo, and Apgar, M.A., 1984, Hydrogeochemical studies at a landfill in Delaware: Studies in Geophysics-Groundwater Contamination, Washington, D.C., National Academy Press, p. 127-138.

Barker, J.F., Tessmann, J.S., Plotz, P.E., and Reinhard, M., 1986, The organic geochemistry of a sanitary landfill leachate plume: Journal of Contaminant Hydrology, 1, p. 171-189.

Bisdorf, Robert J., 1988, Personal communication, Geophysicist, U.S. Geological Survey-Branch of Geophysics, July 18.

Bluemle, John P., 1967, Geology and groundwater resources of Traill County: North Dakota Geological Survey Bulletin 49, Part 1 (North Dakota State Water Commission County Groundwater Studies 10), 34 p.

Cartwright, Keros, 1984, Shallow land burial of municipal wastes: Studies in Geophysics-Groundwater Contamination, Washington, D.C., National Academy Press, p. 67-77.

Cartwright, Keros, and McComas, M.R., 1968, Geophysical surveys in the vicinity of sanitary landfills in northeastern Illinois: Groundwater, v. 6, No. 5, p. 23-30.

Cartwright, Keros, and Sherman, F.B., 1972, Electrical earth resistivity surveying in landfill investigations: reprinted from Proceedings of the 10th Annual Engineering and Soils Engineering Symposium, Moscow, Idaho, p. 77-92.

Cartwright, Keros, Griffin, R.A., and Gilkeson, R.H., 1976, Migration of landfill leachate through unconsolidated porous media: in Advances in Groundwater Hydrology: Proceedings of Symposium held at Chicago, Illinois,

American Water Resources Association, p. 215-227.

Chen, K.Y., and Bowerman, F.R., 1974, Mechanisms of leachate formation in sanitary landfills: in Recycling and Disposal of Solid Wastes, Yen, T.F., ed., Ann Arbor, Michigan, Ann Arbor Science Publishers, Inc., p. 349-367.

Cherry, J.A., 1983, Occurrence and migration of contaminants in groundwater at municipal landfills on sand aquifers: in Environment and Solid Waste, Francis, C.W., Auerbach, S.I., and Jacobs, V.A., eds., Butterworths, Boston, p. 127-147.

Chian, E.S.K., 1977, Stability of organic matter in landfill leachates: Water Research, 2, p. 225-232.

City of Hillsboro Auditor's Office, 1988, Personal communication, Hillsboro City Hall, Hillsboro, North Dakota, October 17.

Clark, T.P., 1975, Survey of groundwater protection methods for Illinois landfills: Groundwater, v.13, No. 4, p. 321-331.

Clayton, Lee, 1980, Geologic map of North Dakota: U.S. Geological Survey, Scale 1:670,000.

Dance, J.T., and Reardon, E.J., 1982, Migration of contaminants in groundwater at a landfill: A case study, 4, cation migration in the dispersion test: Journal of Hydrology, 63, p. 109-130.

Ecology and Environment, Inc., 1987a, Report of sampling activities, Hillsboro landfill, Hillsboro, North Dakota, TDD F08-8704-16, 10 p.

Ecology and Environment, Inc., 1987b, Analytical results report, Hillsboro landfill, Hillsboro, North Dakota, TDD F08-8708-02, 7 p.

Environmental Protection Agency, 1980, Procedures manual for groundwater monitoring at solid waste disposal facilities: Washington, D.C., SW-611, 269 p.

Environmental Protection Agency, 1986a, Census of state and territorial subtitle D non-hazardous waste programs: Washington, D.C., EPA/530-SW-86-039, 178 p.

Environmental Protection Agency, 1986b, Characterization of municipal solid waste in the United States, 1960-2000: Washington, D.C., 63 p.

Environmental Protection Agency, 1986c, Subtitle D study,

phase I report: Washington, D.C., EPA/530-SW-86-054, 239 p.

Freeze, R.A., and Cherry, J.A., 1979, Groundwater: Englewood Cliffs, NJ, Prentice-Hall, Inc., 604 p.

Garland, G.A., and Mosher, D.C., 1975, Leachate effects from improper land disposal: Waste Age, v.6. No. 3, p. 42-48.

Gass, T.E., 1988, Groundwater in the news: Groundwater, v. 26, No.4, p. 540-552.

Giddings, T.M., 1977, The Lycoming County, Pennsylvania, sanitary landfill: State-of-the-art in groundwater protection: Groundwater, v.15, No. 1, p. 5-12.

Griffin, R.A., and Shimp, N.F., 1975, Interaction of clay minerals and pollutants in municipal leachate: Water Reuse, v. 2, p. 801-811.

Griffin, R.A., Cartwright, Keros, Shimp, N.F., Steele, J.D., Ruch, R.R., White, W.A., Hughes, G.M., and Gilkeson, R.H., 1976, Attenuation of pollutants in municipal landfill leachate by clay minerals: Part 1 - column leaching and field verification: Illinois State Geological Survey Environmental Geology Notes No. 78, 34 p.

Griffin, R.A., Frost, R.R., Au, A.K., Robinson, G.D., and Shimp, N.F., 1977, Attenuation of pollutants in municipal landfill leachate by clay minerals: Part 2 - heavy metal adsoption: Illinois State Geological Survey Environmental Geology Notes No. 79, 47 p.

Heigold, P.C., Poole, V.L., Cartwright, Keros, and Gilkeson, R.H., 1985, An electrical earth resistivity survey of the Macon-Taylorville ridged-drift aquifer: Illinois State Geological Survey Circular 533, 23 p.

Hughes, G.M., 1972, Hydrogeologic considerations in the siting and design of landfills: Illinois State Geological Survey Environmental Geology Notes No. 51, 22 p.

Hvorslev, M.J., 1951, Time lag and soil permeability in groundwater observations: Waterways Experiment Station, Corps of Engineers, U.S. Army Bulletin 36, Vicksburg, Mississippi, 47 p.

JCPDS International Centre for Diffraction Data, 1980, Mineral powder diffraction file data book, Swarthmore, Pennsylvania, JCPDS, 1168 p.

Jensen, H.M., and Bradley, Edward, 1963, Groundwater in the

vicinity of Hillsboro, Traill County, North Dakota: North Dakota State Water Conservation Commission, North Dakota Groundwater Studies No. 55, 9p.

Jensen, H.M., and Klausing, R.L., 1971, Geology and groundwater resources of Traill County: North Dakota Geological Survey Bulletin 49, Part 3 (North Dakota State Water Commission County Groundwater Studies 10), 40 p.

Johnson, A.I., 1967, Specific yield - compilation of specific yields for various materials: U.S. Geological Survey Water-Supply Paper 1662-D, 74 p.

Johnson, T.M., and Cartwright, Keros, 1980, Monitoring of leachate migration in the unsaturated zone in the vicinity of sanitary landfills: Illinois State Geological Survey Circular 514, 82 p.

Journal of Hydrology Special Issue, 1983, Migration of contaminants in groundwater at a landfill: A case study, Cherry, J.A., ed., v. 63, No. 1/2, 197 p.

Kehew, A.E., 1977, Land disposal site inventory: North Dakota State Department of Health, Bismarck, North Dakota, 2 p.

Kehew, A.E., and Groenewold, G.H., 1983, Earth resistivity investigations in reclaimed surface lignite mine spoils: North Dakota Geological Survey, Report of Investigation No. 77, 92 p.

Kelly, W.E., 1976, Geoelectric sounding for delineating groundwater contamination: Groundwater, v. 14, No. 1, p. 6-10.

Klefstad, Gilbert, Sendlein, L.V.A., and Palmquist, R.C., 1975, Limitations of the electrical resistivity method in landfill investigations: Groundwater, v. 13, No. 5, p. 418-427.

Kunkle, G.R., and Shade, J.W., 1976, Monitoring groundwater quality near a sanitary landfill: Groundwater, v. 14, No. 1, p. 11-20.

Lindorff, D.E., Feld, J., and Connelly, J., 1987, Groundwater sampling procedures guidelines: Madison, Wisconsin, Wisconsin Department of Natural Resources, PUBL WR-1534-87, 91 p.

Masch, F.D., and Denny, K.J., 1966, Grain size distribution and its effects on the permeability of unconsolidated sands: Water Resources Research, 2, p. 665-667.

Murphy, E.C., and Kehew, A.E., 1984, The effect of oil and gas well drilling fluids on shallow groundwater in western North Dakota: North Dakota Geological Survey, Report of Investigation No. 82, 156 p.

National Oceanic and Atmospheric Administration, 1988, Climatological data and monthly precipitation data: National Weather Service, U.S. Department of Commerce, Bismarck, North Dakota.

Nicholson, R.V., Cherry, J.A., and Reardon, E.J., 1982, Migration of contaminants in groundwater at a landfill: A case study, 6, hydrogeochemistry: Journal of Hydrology, v. 63, no. 1/2, p. 131-176.

North Dakota State Department of Health, Division of Water Supply and Pollution Control, 1988, Draft North Dakota Groundwater Quality Protection Strategy, 89 p.

Palmquist, Robert, and Sendlein, L.V., 1975a, The configuration of contamination enclaves from refuse disposal sites on floodplains: Groundwater, v. 13, No. 2, p. 167-181.

Palmquist, Robert, and Sendlein, L.V., 1975b, A topographic-hydrogeologic model for solid waste landfill siting: Groundwater, v.13, No. 3, p. 260-268.

Pettyjohn, W.A., and Hounslow, A.W., 1983, Organic compounds and groundwater pollution: Groundwater Monitoring Review, v. 3, No. 4, p. 41-47.

Reetz, Rod, 1988, Personal communication, Assistant Director, Division of Chemistry Laboratory, North Dakota State Department of Health, Bismarck, North Dakota, September 7.

Rehm, B.W., Groenewold, G.H., and Peterson, W.M., 1982, Mechanisms, distribution, and frequency of groundwater recharge in an upland area of western North Dakota: North Dakota Geological Survey Report of Investigation 75, 72 p.

Reinhard, M., Barker, J.F., and Goodman, N.L., 1984, Occurrence and distribution of organic chemicals in two landfill leachate plumes: Environmental Science and Technology, v. 18, No. 12, p. 953-961.

Ruffner, J.A., 1985, Climates of the states, volume II: Detroit, Gale Research Co., p. 759-1572.

Saar, R.A., and Braids, O.C., 1983, Chemical indicators of leachate contamination in groundwater near municipal landfills: in Environment and Solid Wastes, Charac-

terization, Treatment, and Disposal, Francis, C.W., and Auerbach, S.I., eds., Butterworths, Boston, p. 315-327.

Schroeder, P.R., Morgan, J.M., Walski, T.M., and Gibson, A.C., 1983a, The hydrologic evaluation of landfill performance model, vol. I, user's guide for version 1: Vicksburg, Mississippi, U.S. Army Engineer Waterways Experiment Station, 100 p.

Schroeder, P.R., Gibson, A.C., and Smolen, M.D., 1983b, The hydrologic evaluation of landfill performance model, vol. II, documentation for version 1: Vicksburg, Mississippi, U.S. Army Engineer Waterways Experiment Station, 107 p.

Schwartz, F.W., and McClymont, G.L., 1977, Applications of surface resistivity methods: Groundwater, v. 15, No. 3, p. 197-202.

Simpson, H.E., 1929, Geology and groundwater resources of North Dakota: U.S. Geological Survey Water-Supply Paper 598, 312 p.

Soiltest, Inc., 1968, Earth resistivity manual: Evanston, Illinois, Soiltest, Inc., 52 p.

Starkey, H.C., Blackmon, P.D., and Hauf, P.L., 1984, The routine mineralogical analysis of clay-bearing samples: U.S. Geological Survey Bulletin 1563, 32 p.

Steiner, R.L., Fungaroli, A.A., Schoenberger, R.J., and Purdom, P.W., 1971, Criteria for sanitary landfill development: Public Works, v. 102, No. 3, 77 p.

Stollar, Robert, and Roux, Paul, 1975, Earth resistivity surveys - a method for defining groundwater contamination: Groundwater, v. 13, No. 2, p. 145-150.

Teller, J.T., and Clayton, Lee, 1983, Glacial Lake Agassiz: The Geological Association of Canada Special Paper 26, 451 p.

Tillotson, Stephan J., 1988, Personal communication, Environmental Scientist, North Dakota State Department of Health-Division of Hazardous Waste Management, June 6.

Water Well Journal, 1974, Groundwater pollution from subsurface excavations, part IX, landfills: v. 28, No. 12, p. 41-44.

Weiss, Samuel, 1974, Sanitary landfill technology: Park Ridge, New Jersey, Noyes Data Corporation, 300 p.

Yazicigil, H., and Sendlein, L.V.A., 1982, Surface geophysical techniques in groundwater monitoring, part II: Ground Water Monitoring Review, v. 2, no. 1, p. 56-62.

Zanoni, A.E., 1972, Groundwater pollution and sanitary landfills-a critical review: Groundwater, v. 10, No. 1, p. 3-16.

Zohdy, A.A.R., and Bisdorf, R.J., 1975, Computer programs for the forward calculation and automatic inversion of Wenner sounding curves, NTIS Publication PB-247 265, 47 p.

Zohdy, A.A.R., and Bisdorf, R.J., 1988, Temporary Wenner sounding data processing and interpretation program, version 1.2.